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The Impact of Aquatic Plants and Their Management Techniques on the Aquatic Resources of the United States: An Overview

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FOREWORD

The protection of aquatic ecosystems from damage caused by toxic organic pollutants and other pest control agents requires that control regulations be formulated on a sound scientific basis. Accurate information describing treatment-response relationships for organisms and ecosystems under varying conditions is required. The EPA Environmental Research Laboratory, Gulf Breeze, contributes to this information through research programs aimed at determining:

- the effects of toxic organic pollutants on individual species and communities of organisms;
- the effects of toxic organics on ecosystem processes and components.

Infestation of aquatic ecosystems by native and exotic plants has increased dramatically in the past 10 years. Herbicides, herbivorous fishes and other organisms that attack plants, and mechanical harvesting are often ineffective in control of nuisance plants, especially for extended periods of time. In addition, secondary effects of control may cause undesireable physical, chemical, and biological characteristics of the aquatic system. This report reviews methods of weed control and their secondary effects. It suggests research needed for weed control that have minimal effect on the quality of water and its use for human needs.

Henry F. Enos

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PREFACE

Aquatic systems in the United States have been studied for a long time, and there is a voluminous quantity of information on the impact of aquatic plants and their management techniques on the aquatic environment. It is beyond the scope of this paper to summarize each study. We have, therefore, attempted to select those studies which have documented various impacts of aquatic plants or their management techniques on the aquatic environment. Inclusion of any particular study in this review paper does not constitute an endorsement of the results or conclusions. We have not examined data for errors and have made no judgment as to quality of the studies. We report only what is currently accepted in the published scientific literature. Because results and conclusions from different studies often conflict, readers of this paper should not accept the results or conclusions of any particular study until additional scientific research clarifies the discrepancies.

Throughout this paper there are many subjects which we have discussed in the most general of terms. In these cases, we have not provided factual information because information is lacking and most studies lack data to support the generalizations made. In other instances, there may be voluminous amounts of physical and biotic data available. However, because there was often no rationale for collection of the data or appreciation of interrelationships between physical and biotic factors, most of these data have not been analyzed. Without our own analysis, all we could possibly do is list the data in tables which would not provide much additional insight beyond our general description of the reported results.

As an additional comment, this review paper has made us aware of the large amounts of information now available. We hope our review of the literature on the impact of aquatic plants and their management techniques on water quality and the aquatic environment will be of use to water resource managers. We must, however, caution all water resource managers about attempting to use results from studies cited in this paper to manage aquatic systems that differ from the studied systems. There is an urgent need for studies on large numbers of aquatic systems in order to document quantitative patterns in the behavior of aquatic systems. Once these quantitative empirical relationships have been obtained, water resource managers will be able to manage our nation's aquatic resources with greater confidence. Until these empirical relationships are available, we recommend caution before a particular management strategy is adopted.

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ABSTRACT

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INTRODUCTION

Excessive growth of aquatic plants often seriously interferes with many domestic, agricultural, industrial, and recreational water uses. For this reason, a number of diverse management techniques have been developed that can reduce the growth of aquatic plants (Dunst et al. 1974). In the past, these techniques were often used without knowledge of their long-term impact on the aquatic environment, and a technique was considered good if it controlled aquatic plants. Water, however, is becoming an increasingly valuable resource throughout the United States and many user groups are now expressing concern about the impact of aquatic plant management techniques on the aquatic environment. Because of this concern, there have been increased demands for development of effective, yet environmentally safe aquatic plant management programs.

Formulation of aquatic plant management programs for different aquatic systems is extremely difficult. It is often very difficult to obtain an accurate assessment of the extent of an aquatic plant problem because of the wide range of public and private waters affected and the numerous overlapping governmental jurisdictions responsible for managing aquatic plant problems. If aquatic plant management programs are not to become mere cosmetic treatment programs, information is needed on the factor or factors that control aquatic plant growth. This information, however, is often unavailable. Finally, information is needed on the impact of various aquatic plant management techniques on the aquatic environment. Study of the aquatic environment is not new (Forbes 1887; Apstein 1896; Kofoid 1903; and others), and extensive literature exists concerning the general limnology of different aquatic systems, the limnological roles of aquatic plants, and the impact of different management techniques on the aquatic environment (Gessner 1955, 1959; Hutchinson 1957, 1967, 1975; Sculthorpe 1967; Dunst et al. 1974; Wetzel 1975; and others). However, very few patterns in the behavior of aquatic systems have been quantitatively documented. Thus it is very difficult to predict the short-term and long-term impact of aquatic plant management programs.

In this review paper, we attempt to provide a general assessment of the aquatic plant problem in the United States. We also present information on what is currently known about the impact of aquatic plants and management techniques on water quality. Due to the complexity of aquatic systems, we also discuss the overall impact of aquatic plants and management techniques on the aquatic environment. We include this information because other components of the aquatic environment which are affected either directly or indirectly by aquatic plants can also affect water quality and because without this information it is impossible to provide a balanced perspective of the total impact of aquatic plants and techniques for their management on the water resources of the United States. Finally, after reviewing the information currently available, we identify what we consider to be major research needs if effective, environmentally safe aquatic plant management programs are to be developed.

PROBLEM AQUATIC PLANTS

Algae

Algae are found in nearly all waters located throughout the United States. In 1972, the United States Environmental Protection Agency (EPA) initiated the National Eutrophication Survey to investigate the nationwide threat of accelerated cultural eutrophication to freshwater lakes and reservoirs. This survey collected physical, chemical, and biological data on more than 800 lakes and reservoirs throughout the contiguous United States. Figure 1 shows the relationship between algal biomass as measured by chlorophyll *a* concentrations and total phosphorus concentrations for the sampled lakes and reservoirs. If chlorophyll *a* values above 10 mg/m^3 and total phosphorus values above 20 mg/m^3 are used to demarcate eutrophic lakes and reservoirs with algal problems, algal problems occur in many of the nation's lakes and reservoirs (Figure 1).

Research on natural lakes and reservoirs (Edmondson 1961; Sakamoto 1966; Vollenweider 1968; Dillon and Rigler 1974a; Schindler 1975; Jones and Bachmann 1976; Canfield 1979) has shown a strong correlation between algal biomass as measured by chlorophyll *a* concentrations and total phosphorus concentrations (Figure 2). This research strongly suggests that phosphorus is the element most likely limiting algal biomass. Whole-lake experiments by Schindler (1975) have further shown that reduction in phosphorus input to lakes will significantly reduce algal biomass. For this reason, there has been a nationwide effort to reduce phosphorus inputs to lakes and reservoirs.

Nutrient reduction programs have been successful on a number of lakes (Michalski and Conroy 1973; Schindler 1975), with Lake Washington (Edmondson 1961, 1966, 1969, 1970, 1972a, 1972b) being a clasic example of the benefits of controlling nutrient inputs. Other lakes with a long history of high phosphorus inputs, however, have recovered very slowly with reductions in phorphorus inputs (Ahlgren 1972; Bjork 1972; Larsen et al. 1975). While Shapiro (1979), Canfield (1979), and others suggest that factors other than phosphorus, such as zooplankton and sediments, affect algal biomass, reduction of plant nutrient inputs, especially phosphorus, is an important first step in management of algal problems in the United States. Additional research on lake biology, internal recycling on plant nutrients, and algal population dynamics, however, will be required in order to determine how to manage algal problems in waters where it is impossible to reduce nutrient input.

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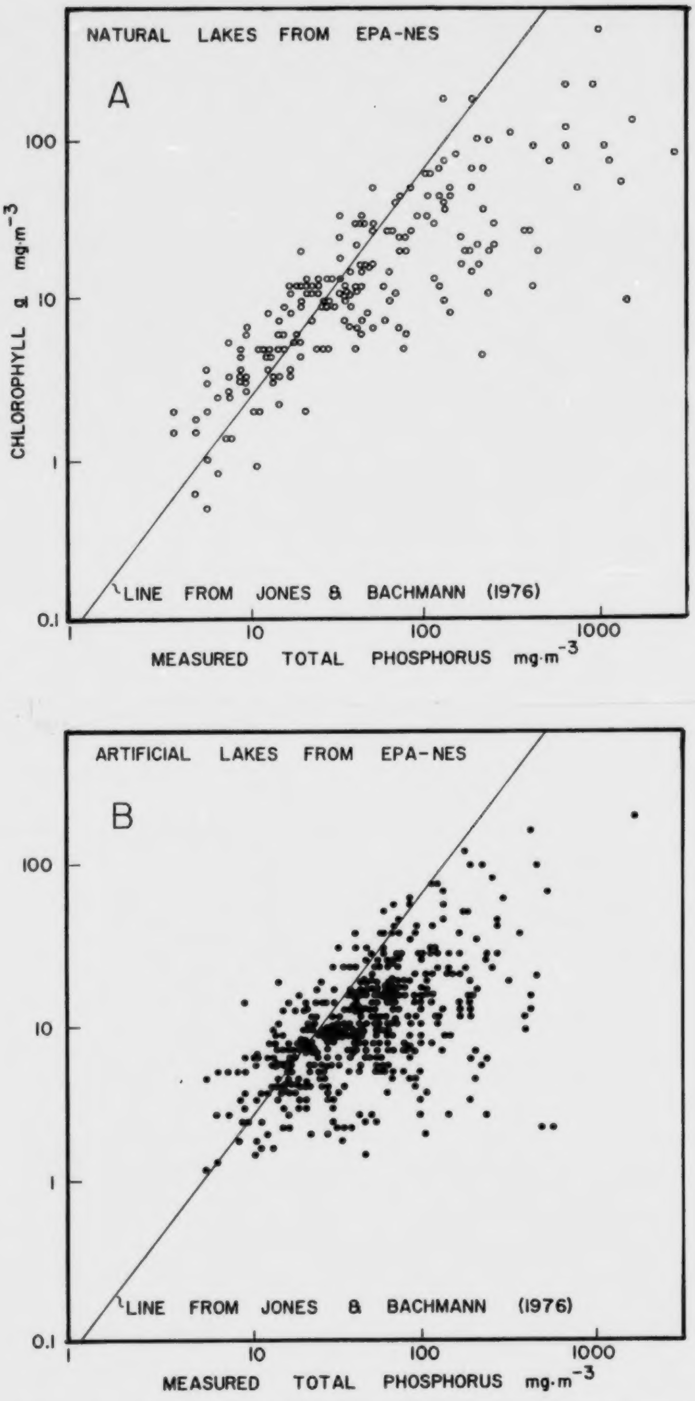

Figure 1. Relationship between chlorophyll a concentrations and measured total phosphorus concentration for the EPA - NES natural and artificial lakes. From Canfield, 1979.

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Italicize Isoetes Nitella Potamogenton

Table 1.
Classification of English lakes on the basis of their physical characters and rooted vegetation. From Pearsall, 1929.

Lake	Drainage System per cent cultivable	Lake Shore* per cent Rocky	Relative transparency of water	Percent of submerged vegetation		
				Isoetes	*Nitella*	*Potamogenton*†
Wastwater	5.2	73	9.0	49	36	½
Ennerdale	5.4	66	8.3	35	48	½
Buttermere	6.0	50	8.0	40	40	1
Crummock	8.0	47	8.0	48	26	2
Hawes Water	7.7	25	5.8	5	71	5
Derwentwater	10.0	33	5.5	31	42	6
Bassenthwaite	29.4	29	2.2	42	3	3
Coniston	21.8	27	5.4	34	9	30
Ullswater	16.6	28	5.4	34	15	35
Windermere	29.4	28	5.5	9	40	38
Esthwaite	45.4	12	3.1	2	26	56

* To a depth of 30 ft.
† Including *Naias* and *Elodea* in small quantities.

Native Aquatic Macrophytes

Native aquatic macrophytes are also found in nearly all waters in the United States and may produce nuisance growths which require control. To the best of our knowledge, there has been no study similar to the National Eutrophication Survey which details the extent of native aquatic macrophyte problems. Reports, however, of either localized or extensive native macrophyte problems caused by emergent (*Typha* sp., *Polygonum* sp., *Eleocharis* sp., and others), submergent (*Potomogeton* sp., *Utricularia* sp., *Elodea* sp., *Cabomba* sp., and others), floating-leafed (*Nelumbo lutea*, *Nuphar* sp. and others), and floating (*Lemna* sp., *Spriodela* sp., *Wolffia* sp., and *Wolffiella* sp.) aquatic plants can be found.

Unlike for algae, there is a general lack of information on the factor or factors that limit growth of aquatic macrophytes. Early works by Pearsall (1921, 1922) have suggested that edaphic factors, particularly the coarseness of the bottom, determine distribution of different types of aquatic plants in English lakes (Table 1). Pearsall also noted that finer bottom silts generally were associated with higher nutrient concentrations and generally support abundant macrophyte growth. Although these data would seem to suggest that plant nutrients control growth and distribution of aquatic macrophytes and that increased nutrient inputs resulting from human activities might be responsible for increased native macrophyte growth, there are very little data to support this hypothesis. There also have been no studies to determine if reductions in nutrient inputs will cause a corresponding reduction in growth of aquatic macrophytes. Additional research is urgently needed in these areas.

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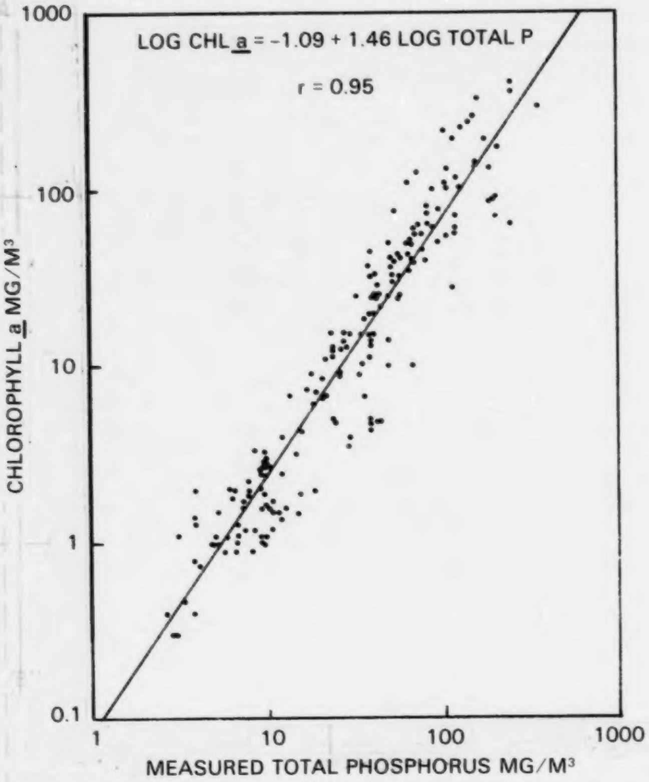

Figure 2. Relationship between summer levels of chlorophyll a and measured total phosphorus concentration for 143 lakes. From Jones and Bachmann, 1976.

Non-native Aquatic Macrophytes

Water hyacinth (Eichhornia crassipes), and hydrilla (Hydrilla verticillata), Eurasian watermilfoil (Myriophyllum spicatum) and alligatorweed (Alternanthera philoxeriodes) are all introduced plant species that have or are currently causing serious aquatic weed infestations in the United States. As with native aquatic macrophytes, there has been no national study to document the extent of the non-native aquatic macrophyte problem. Since its introduction into Lousisiana in 1908, water hyacinth has become distributed throughout the South and parts of California (Figure 3). Hydrilla, which is currently a major aquatic plant problem, has spread since its introduction in 1960 (Figure 4) throughout Florida (Figure 5) where it now covers over 250,000 ha of Florida's fresh water. Hydrilla is now present in Alabama, Mississippi, Georgia, South Carolina, Louisiana, Texas, California, and Iowa (Figure 6) and may pose a serious threat to other states. Eurasian watermilfoil is currently the most widespread aquatic macrophyte that causes problems in the United States (Figure 7). The area of most rapid colonization seems to be in the lower Great Lakes-St. Lawrence River area, but severe infestations have been found in reservoirs of the Tennessee Valley Authority (TVA) and some of the lakes and reservoirs in Washington.

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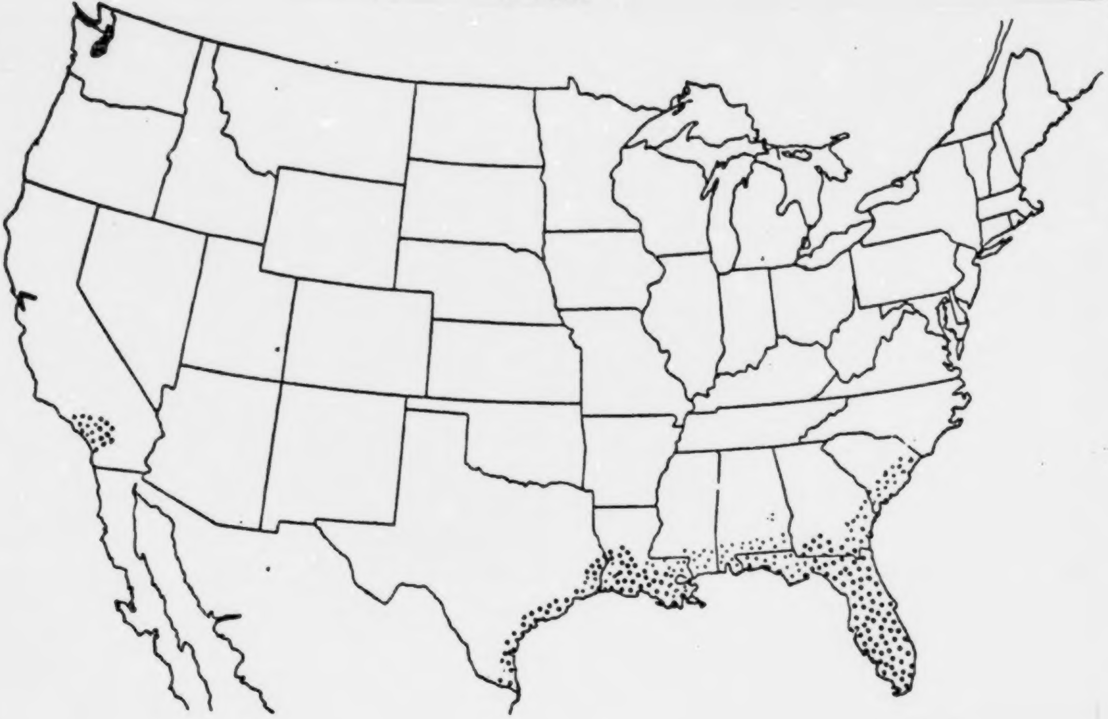

Figure 3. Distribution of water hyacinths in the United States, 1978.

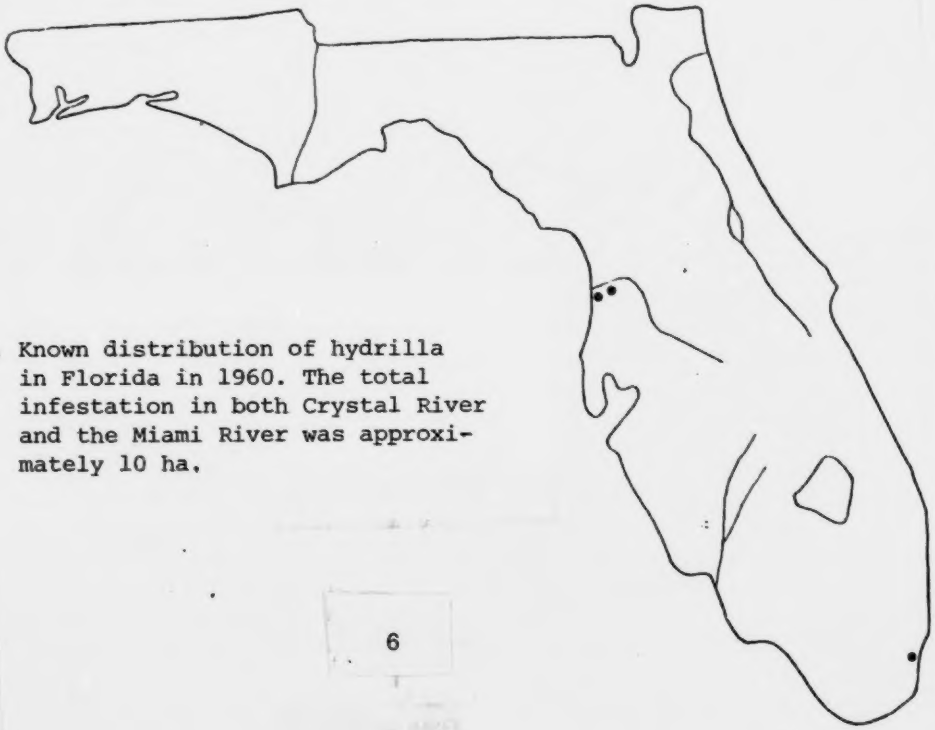

Figure 4. Known distribution of hydrilla in Florida in 1960. The total infestation in both Crystal River and the Miami River was approximately 10 ha.

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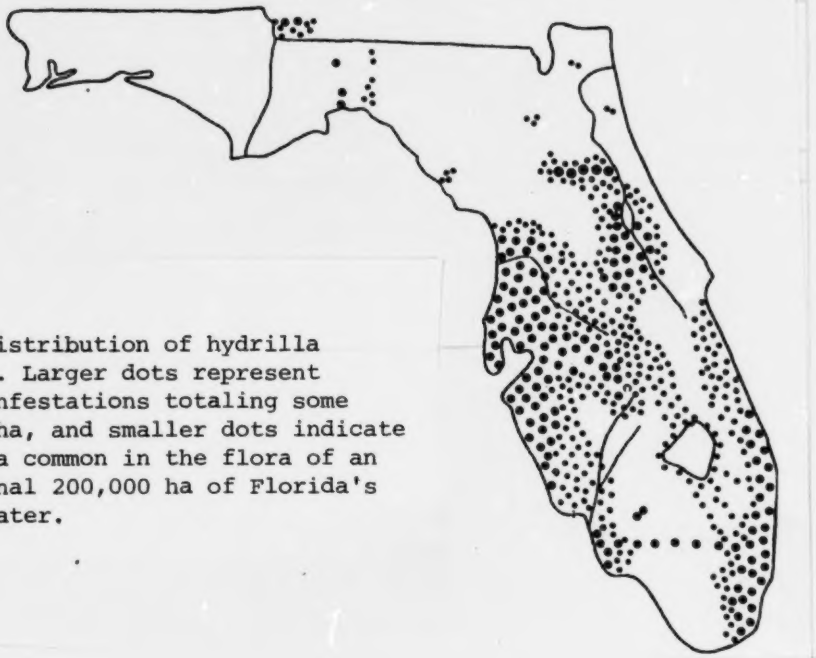

Figure 5. Known distribution of hydrilla in 1978. Larger dots represent dense infestations totaling some 40,000 ha, and smaller dots indicate hydrilla common in the flora of an additional 200,000 ha of Florida's fresh water.

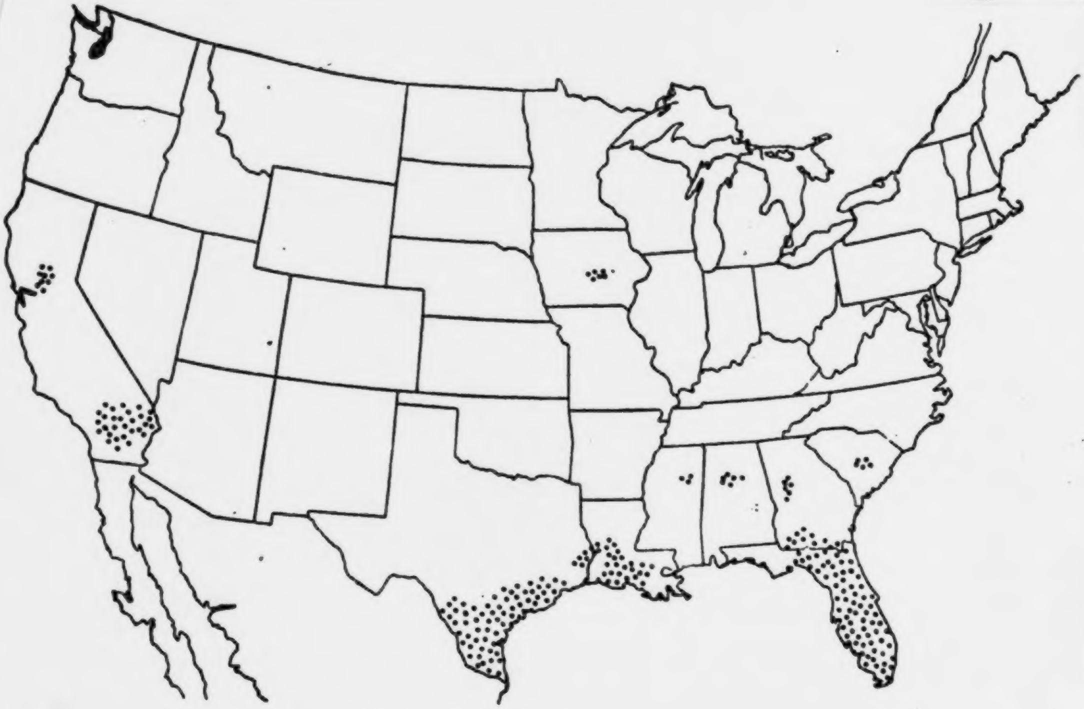

Figure 6 . Distribution of hydrilla in the United States, 1978.

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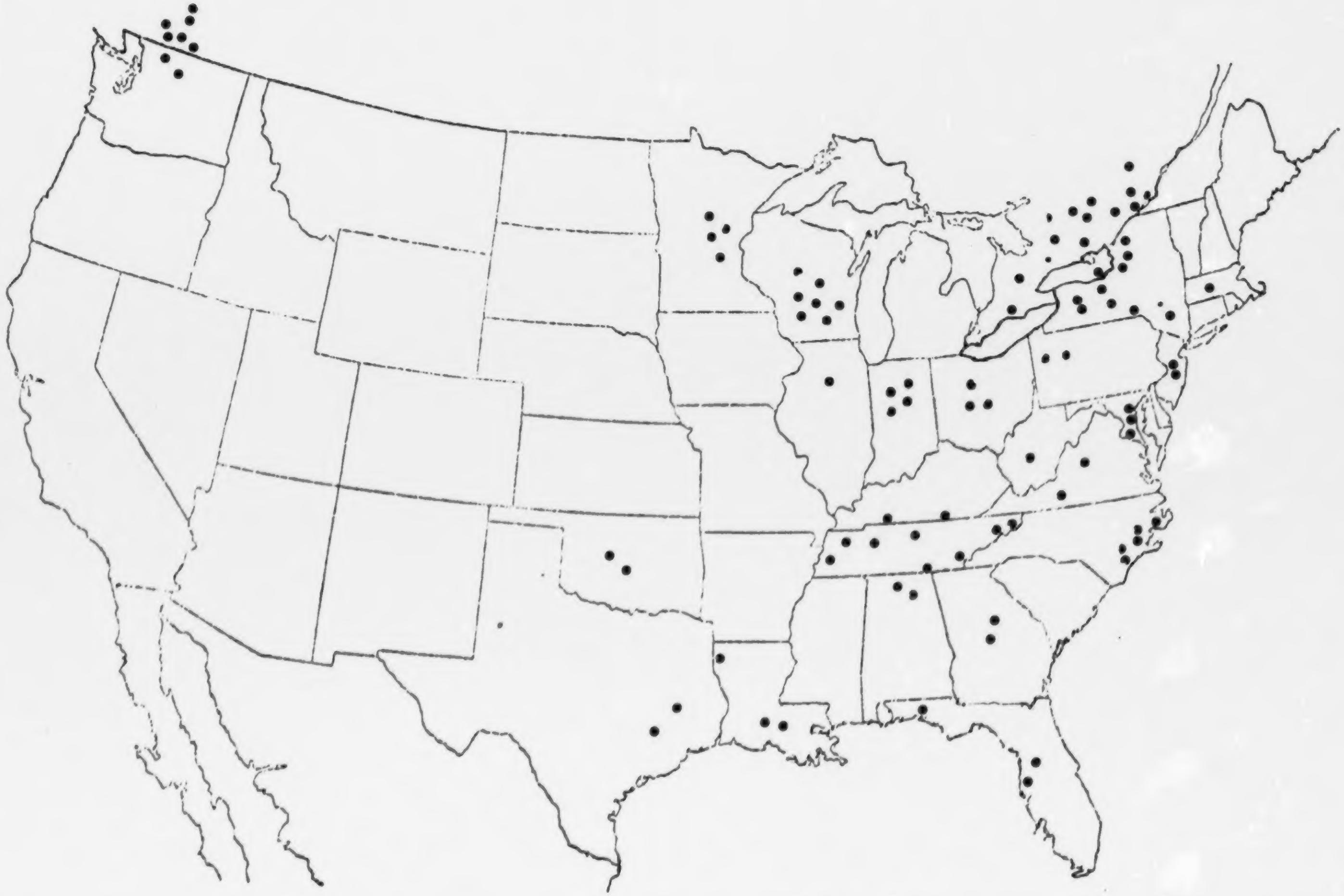

Figure 7. Distribution of Eurasian watermilfoil in the United States, 1978.

There are practically no data on factors that might limit growth of these non-native aquatic macrophytes in the United States. Alligator-weed, which once caused serious problems, is being controlled by the alligator-weed flea beetle (*Agasicles hygrophila* Selman and Vogt) which was introduced into the United States. Water hyacinths are currently being managed at tolerable levels through the use of the herbicide 2,4-D, but in backwater areas and little-used canals where it is often impossible or too costly to spray, water hyacinths tend to cover the surface. Herbicides are also being used in an attempt to control hydrilla and watermilfoil, but with difficulty because the plants are submersed. In the case of hydrilla, the plant can reproduce vegetatively (Haller 1976) and produce underground tubers which are unaffected by most control methods (Miller et al. 1976). It has been speculated that nutrient enrichment of the nation's waters due to human activities might be contributing to the rapid spread of aquatic weeds. While nutrients no doubt affect growth of these plants, cultural eutrophication probably is not the major cause of non-native aquatic macrophyte infestations. What we are probably witnessing is the invasion of new territory by competitively superior plants. For example, Van et al. (1976) showed that hydrilla had a lower light requirement for photosynthesis than some native plants. They suggested this provided hydrilla with a competitive advantage in the uptake of carbon dioxide, a necessary plant nutrient. Further research will be needed to determine if there are natural factors that can be managed to limit the growth of non-native aquatic macrophytes.

Impact of Aquatic Plants on the Aquatic Environment

Impact of Aquatic Plants on Physical Characteristics

Optical Properties. A major water quality parameter that is readily evaluated by the general public and significantly affected by aquatic plants is water clarity. The development of excess plankton algae can significantly reduce water clarity. Recent studies on natural lakes (Edmondson 1972a,b; Bachmann and Jones 1974; Dillon and Rigler 1974a; Oglesby and Schaffner 1975; Jones and Bachmann 1978) have shown a hyperbolic relationship between algal biomass as estimated by chlorophyll *a* concentrations and water clarity as measured with Secchi disc (Figure 8). Using data from a large number of lakes (Figure 9), Jones and Bachmann (1978) showed that water clarity as measured with a Secchi disc (SD) could be predicted from chlorophyll *a* (Chl *a*) concentrations by the following equation:

$$\text{Log SD} = 0.807 - 0.549 \text{ Log } (\text{Chl } \underline{a} + 0.03) \qquad (1)$$

These relationships (Figures 8 and 9) clearly show that significant improvements in water clarity cannot be expected until chlorophyll *a* concentrations are reduced below 10 mg/m^3. With these relationships, water resource managers can predict not only water clarity, but the general public's response to either increases or reductions in algal concentration.

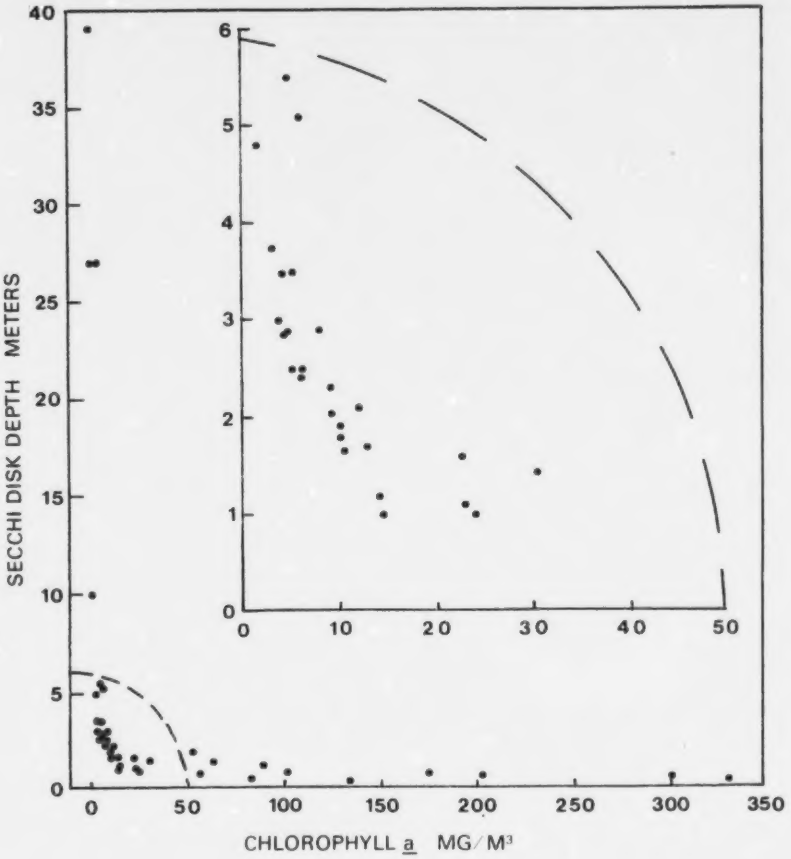

Figure 8. Relationship between mean Secchi disc transparencies for July and August and the mean July-August chlorophyll a concentrations for 16 lakes. From Bachmann and Jones, 1974.

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Not all aquatic plants, however, reduce water clarity. A number of researchers have noted that dense growths of aquatic macrophytes seem to inhibit development of phytoplankton populations and are generally associated with clear water (Kofoid 1903; Schreiter 1928; Hasler and Jones 1949; Hogetsu et al. 1960; Stangenberg 1968; Goulder 1969). If aquatic macrophytes actually inhibit phytoplankon development, excellent water clarity associated with dense macrophyte growths probably results from reduction in algal numbers. Irwin (1945) also suggested that aquatic macrophytes enhance reduction of clay turbidities by reducing charges on the clay particles and thus improving water clarity. Aquatic macrophytes might also contribute to improved water clarity by reducing resuspension of bottom sediments as Pond (1903) and Klugh (1926) suggested. There are, however, no data relating aquatic macrophyte biomass to water clarity in lakes, thus it is very difficult to predict what level, if any, that aquatic macrophyte biomass will contribute to a significant improvement or reduction in water clarity. Additional research is especially needed to ascertain quantitatively the relationship between aquatic macrophyte biomass and algal biomass as available data clearly show algae sigificantly affect water clarity.

Although water clarity is readily evaluated as an index of water quality and clear water has a greater aesthetic value than turbid water, the general public also uses water color as an index of water quality. Phytoplankton are often responsible for imparting dull green, blue-green, or yellowish-brown colorations to water (Naumann 1922; Hutchinson 1957; Wetzel 1975). Occasionally, populations of the genera *Euglena*, *Haematococcus*, or *Glenodium* may impart a blood-red coloration to the water (Klausner 1908; Huber-Pestalozzi 1936; Wetzel 1975). To the best of our knowledge, relationships between algal biomass and water color are based strictly on observational data. There seem to be very little quantitative data that can be used to determine the algal biomass at which color becomes noticeable. If quant- itative data could be obtained, it might be possible to develop an empirical relationship between algae and water color that would be of use to water resource managers in predicting response of the general public to increases or reductions in algae. This type of information is important to water resource managers because, regardless of what the physical, chemical, or biological limnological data may show, the general public must perceive a change or else often will assume that no change has occurred.

Aquatic plants also can significantly affect other optical properties of the aquatic environment. The relationship between Secchi disc depth and algal biomass as measured by chlorophyll *a* concentrations clearly shows that an increase in algal biomass reduces depth of light penetration (Figure 9). Studies by Beeton (1958), Stepanek (1959), and Tyler (1968) have shown that the Secchi disc depth represents from 1 to 15% of the surface light transmitted. Studies by Oosting (1933), Gessner (1955), Szumiec (1961), and Haller and Sutton (1975) have shown that the amount of light entering various depths of the aquatic environment can be significantly affected by dense growths of aquatic plants (Figure 10). Ultsch (1973) observed that dense mats of water hyacinths can effectively block all light from entering the aquatic environment.

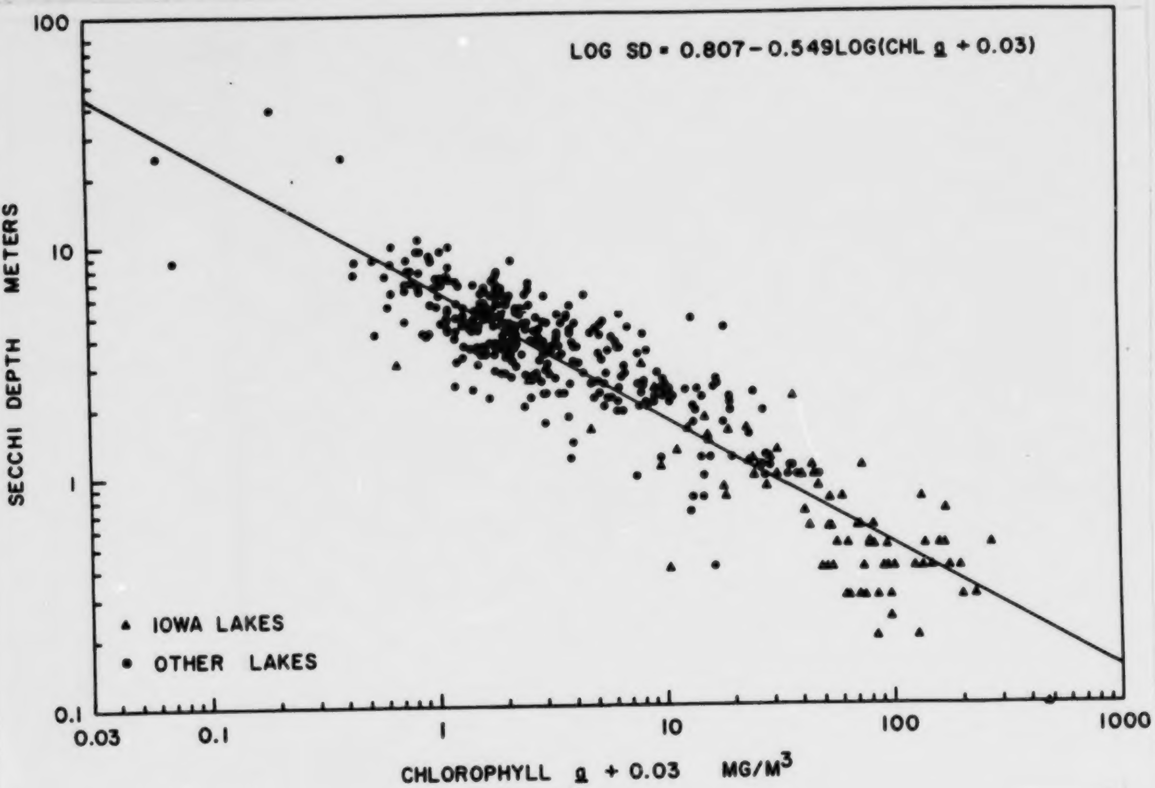

Figure 9. Double logarithmic plot of Secchi disk depths against average chlorophyll a concentrations. Data for non-Iowa lakes were taken from the literature (Bachmann and Jones, 1976, Dillon and Rigler, 1975, Oglesby and Schaffner, 1975) and from reports of lake self-help projects in Ontario and Michigan. The addition of the constant 0.03 to the chlorophyll values prevents the calculated Secchi disk values from approaching infinity as the chlorophyll levels approach zero. From Jones and Bachmann, 1978.

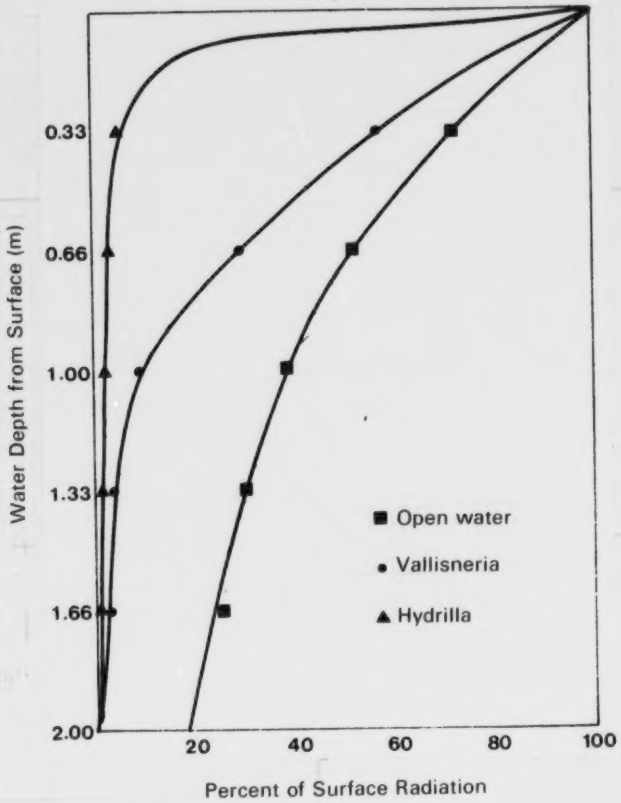

Figure 10. Light penetration through water and plant communities in Rodman Reservoir. Light measurements in the hydrilla community were made through small (20 cm^2) openings in the canopy. From Haller and Sutton, 1975.

Aquatic plants can directly influence the quantity of light entering the aquatic environment and also indirectly influence its quality. Studies (Wetzel 1969a; 1969b; Fogg 1971; Hellebust 1974; and others) have strongly suggested that dissolved organic compounds are released by aquatic plants. James and Birge (1938) showed that increasing concentrations of dissolved organic compounds as measured by water color not only reduce the depth to which light is transmitted, but shift absorption selectivity (Table 2). Though most of the dissolved organic compounds studied by James and Birge (1938) were probably derived plant material produced within the watershed, their data strongly suggest that aquatic plants can influence not only the quantity, but also the quality of light entering the aquatic environment. Because the quantity and quality of light entering water influences photosynthesis, and hence production by various aquatic plants, research should be conducted to determine how quantitative and qualitative changes affect distribution, abundance, and production of aquatic plants.

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Table 2.
Percentile absorption of light of different wavelengths by one meter of lake water, settled of particulate matter, of several Wisconsin lakes of progressively greater concentrations of organic color[a]. From Wetzel, 1975.

Wavelength (nm)	Distilled Water	Crystal Lake	Lake Mendota	Alelaide Lake	Mary Lake	Helmet Lake
800	88.9	89.9	90.5	92.4	91.7	93.2
780	90.2	91.3	91.9	93.5	93.0	94.5
760	91.4	93.5	92.6	94.5	94.8	96.0
740	88.5	89.3	91.5	92.7	93.0	96.2
720	64.5	67.6	71.0	78.0	78.0	86.9
700	45.0	50.4	49.7	66.3	70.7	82.5
685	38.0	45.2	42.2	65.7	71.7	86.6
668	33.0	40.3	36.8	65.0	72.3	88.0
648	28.0	37.0	31.9	64.5	75.2	91.2
630	25.0	34.4	28.9	65.8	77.8	94.0
612.5	22.4	32.1	26.3	66.8	80.3	96.0
597	17.8	27.5	22.5	67.0	83.2	97.6
584	9.8	22.0	17.6	67.1	85.7	98.2
568.5	6.0	19.3	14.0	67.6	88.5	98.6
546	4.0	19.2	13.5	70.9	91.6	99.3
525	3.0	19.8	14.1	74.5	94.8	—
504	1.1	20.7	15.2	81.0	97.4	—
473	1.5	21.7	21.7	88.6	99.4	—
448	1.7	23.8	27.8	92.2	—	—
435.9	1.7	24.4	31.0	95.2	—	—
407.8	2.1	28.1	44.3	99.0	—	—
365	3.6	40.0	80.0	—	—	—
Color Scale (Pt units)	0	0	6	28	101	264

[a]Selected data from James and Birge, 1938.

Thermal Properties. Absorption of solar energy by water is influenced by a number of physical, chemical, and, under certain conditions, biological characteristics of the water (Wetzel 1975). Because water temperatures may influence various physical, chemical, and biological characteristics of the aquatic environment, it is important to understand the effect of aquatic plants on the thermal properties of different aquatic systems. Oosting (1933) noted that water temperatures were frequently lower in beds of floating or floating-leafed aquatic macrophytes than in open water areas. Dvorak (1970) also found consistently lower temperatures within stands of the emergent Glyceria than in open water. His data showed that water temperatures averaged about 17 C within the Glyceria stands but about 21 C in open water. Hotchkiss (1941) observed that surface water over submersed aquatic beds heated very rapidly to relatively high temperatures. He suggested that surface heating resulted because aquatic macrophytes restricted water movements and thus prevented mixing of warm water with cooler water.

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Although there are data that indicate aquatic plants can affect thermal properties of aquatic environments, there are no data that indicate how important their role is in aquatic systems of differing sizes or depths. It would be interesting to know how aquatic plants affect the total heat budget of different aquatic systems. It also has been speculated that aquatic plants, by affecting temperatures, can affect the behavior of aquatic organisms such as fish. There are, however, very little data on which to judge just how important temperature modifications by aquatic plants are to other aquatic organisms. Because temperature is an important water quality parameter, studies are needed to determine quantitatively how different levels of aquatic plants affect the thermal properties of different aquatic systems.

Water Movements. The movement of water is extremely important in the aquatic environment as it can transport plant nutrients and organic matter, add oxygen to water through surface aeration, affect silt deposition and erosion sites, and either directly or indirectly affect various water quality parameters. Direct measurements of the impact of aquatic plants on water movements have been few. Studies have shown that aquatic plants in irrigation and drainage canals can impede the flow of water (Stephens et al. 1957; Timmons 1967). Stephens et al. (1957) showed that water hyacinths reduced flow by 40%, while submersed aquatic macrophytes reduced flow by 97% in extreme cases. Observations of lakes by various researchers (Oosting 1933; Hotchkiss 1941; Seabrook 1962; Dvorak 1970; Unni 1972) have suggested that aquatic plants, especially aquatic macrophytes, can significantly reduce water movements. The lack of data, however, makes it very difficult to determine the magnitude of water movement reductions even though some researchers have suggested plants can reduce water movements to the point of stagnation.

Water flow and circulation can affect nutrient dynamics, the transport and final deposition of sediments and associated toxicants, and a host of other parameters. Therefore, detailed studies are needed to quantify the impact of various aquatic plants on water movements. At the present time, there are no quantitative data to indicate at what levels various types of aquatic plants affect water movements or how significant these effects might be to the functioning of aquatic systems. Quantitative data should be collected from different aquatic systems to determine if these are general behavior patterns which would permit the prediction of the impact of different levels of aquatic plants on water movements.

Water Balance. The extent and rate of evaporative losses from the aquatic environment are affected by many factors. It has long been recognized, however, that transportation by floating, floating-leafed, and emergent aquatic macrophytes (Gessner 1959; Wetzel 1975) can greatly increase evaporative losses. For example, studies by Penfound and Earle (1948), Timmer and Weldon (1967), and Holm et al. (1969) have shown that evaporative water loss due to water hyacinths was 3.7 to 7 times greater than evaporative loss from open water (Table 3).

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Table 3.
Transpiration for water hyacinth in relation to evaporation. From Penfound and Earle, 1948.

Date	Conditions	Total transpiration, Milliliters	Evaporation, Milliliters	Transpiration, Evaporation
June 17-June 19	Clear	8,650	1,900	4.5
June 20-June 28	Cloudy, rain	27,400	10,500	2.6
June 20-July 2	Clear	11,700	2,800	4.2
July 3-July 9	Cloudy, rain	26,900	13,200	2.0
July 10-July 20	Clear, except 13th, 14th	31,700	4,800	6.6
Total	Clear, 13 days; Rain, 21 days	106,350	33,200	3.2

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Annual water loss from irrigation canals due to evapotranspiration by aquatic plants may be considerable (Timmons 1960). Hotchkiss (1941) suggested that transpiration by aquatic plants was especially important in drying of shallow aquatic systems, thus enhancing establishment of terrestrial plants.

Although it is recognized that transpiration by aquatic plants increases the rate of water loss, there is very little information on loss rates for different types of aquatic plants. Further, to the best of our knowledge, water loss rates have not been determined for different quantities of aquatic plants. Research is needed in these areas if water resource managers are to decide rationally what types of aquatic plants and how many aquatic plants should be allowed to grow in various aquatic systems. This information is especially needed in regions of the United States where irrigation is important or surface water supplies are limited.

Basin Morphometry. Basin morphometry may have important effects on nearly all major physical, chemical, and biological characteristics of the aquatic environment (Thienemann 1927; Hutchinson 1938; Rawson 1952, 1955, 1956; Patalas 1961; Hayes and Anthony 1964; Sakamoto 1966). For example, studies by Rawson (1955) have shown that the average standing crop of plankton (Figure 11), the average weight of bottom fauna (Figure 12), and the average long-term commercial fish production (Figure 13) increase as mean depth decreases in lakes. Aquatic plants may have a major long-term indirect impact on aquatic systems by filling in of basins through precipitation of calcium carbonate (Welch 1952; Otsuki and Wetzel 1974; Wetzel 1975), entrapment of inflowing sediments (Hotchkiss 1941), and accumulation of their remains (Wilson 1945; Wetzel 1975). The role aquatic plants play in the modification of basin morphometry, however, is often overlooked by water resource managers when they attempt to evaluate the impact of aquatic plants on the aquatic environment.

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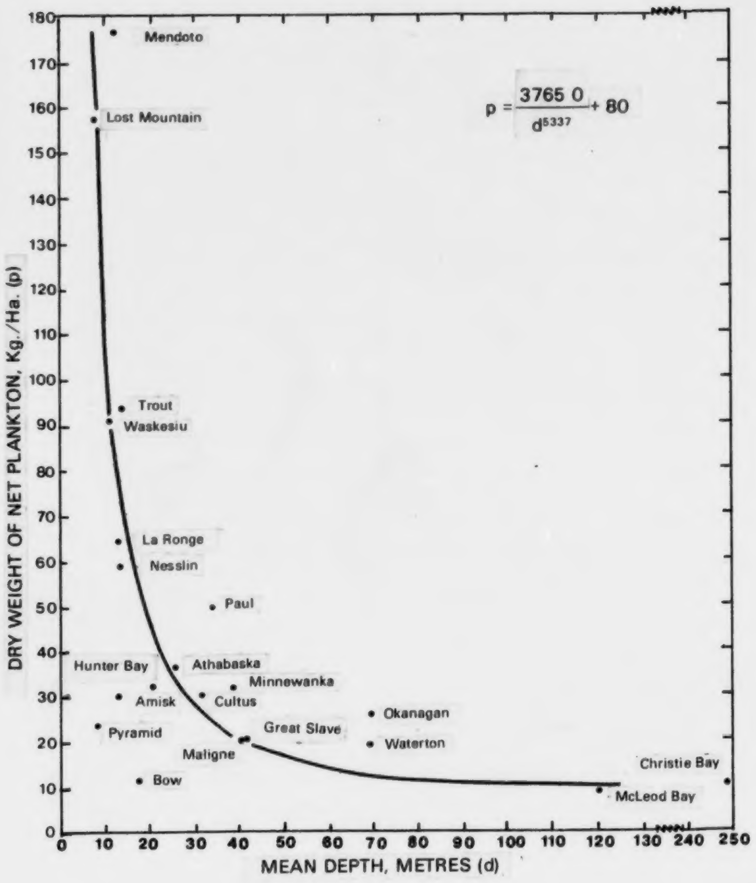

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Figure 11. Mean depth and the average standing crop of plankton in twenty lakes. From Rawson, 1955.

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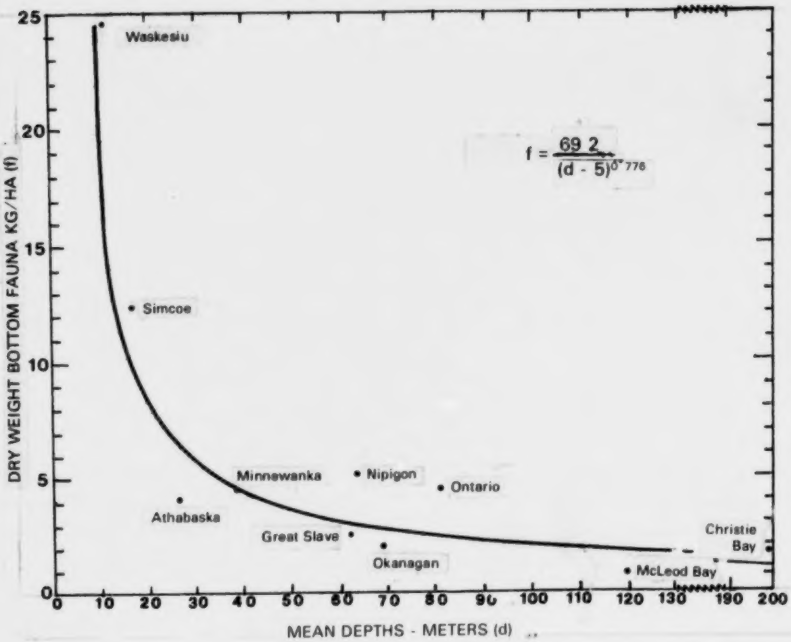

Figure 12. Mean depth and the average weight of bottom fauna in twelve lakes. From Rawson, 1955.

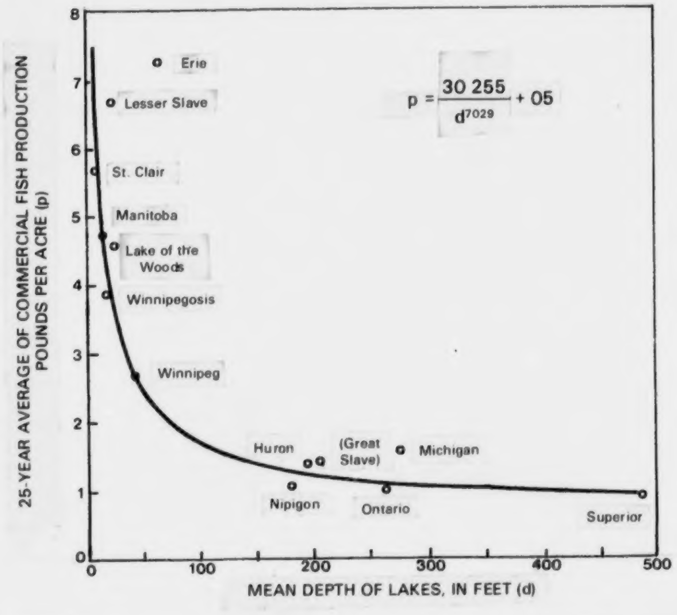

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Figure 13. Mean depth and the long-term average commercial fish production in twelve lakes. From Rawson, 1955.

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Water resource managers often consider aquatic plants to be detrimental to an aquatic system because they contribute to the infilling of the basin and eventually to destruction of the system. Rawson's work, however, strongly suggests that in some aquatic systems, reduction in mean depth resulting from the presence of aquatic plants may have beneficial results, such as increasing fish production. The infilling of an aquatic basin, however, often does not occur uniformly across the whole basin. Pond (1903) and Klugh (1926) suggested that establishment of rooted aquatic macrophytes retards erosion of shallow water sediments to deeper waters. Wilson (1938), Hotchkiss (1941), and Welch (1952) suggested that stablization of bottom sediments enhanced the buildup of littoral zones. The importance of littoral zones as sites of nutrient recycling and as nursery areas for various organisms has long been recognized. The special significance of littoral development has been discussed by Strøm (1928), Alsterberg (1930), Strøm (1930), and Fee (1979).

Although aquatic plants may contribute to the eventual destruction of the basin, it should also be recognized that in some cases aquatic plants are responsible for the formation and preservation of aquatic systems. Murray (1910) suggested that aquatic plants, by blocking drainage channels, were responsible for formation of many aquatic systems in low-lying regions in the tropics and arctic. Penfound and Earle (1948) and Welch (1952) noted that organic dams formed by aquatic plants could significantly increase water elevations in existing aquatic systems. Parker and Cooke (1944) further suggested that aquatic plants, by blocking drainage channels, were responsible for preservation of Lake Okeechobee, Florida. Because aquatic plants can reduce the power of waves (Pond 1903; Oosting 1933; Welch 1952), they can reduce shoreline erosion which subsequently reduces accumulation of sediment in deeper waters.

The impact of aquatic plants on basin morphometry and subsequent direct and indirect effects on physical, chemical, and biological characteristics of the aquatic environment are probably not considered by water resource managers, because many of the changes in basin morphometry occur over long periods of time. Long-term management of water resources, however, will require that the impact of aquatic plants on basin morphometry be considered. At the present time, it is very difficult for water resource managers to predict the impact of changes on basin morphometry. Except for Rawson's work on the importance of lake mean depth, there are no general quantitative relationships that can be used to predict the impact of changes in basin morphometry on physical, chemical, or biological characteristics of the aquatic environment. Even Rawson's 1955 work is based on data from only a few lakes. Additional research will be needed to verify Rawson's relationships. Research should also be conducted to determine the relationships of volume-depth distributions to characteristics of the aquatic environment. Finally, research is needed to determine the rate at which plant remains accumulate and if accumulation rates vary for different types and abundances of aquatic plants. Without this information, water resource managers will be unable to predict the rate at which aquatic plants modify basin morphometry.

Impact of Aquatic Plants on Chemical Characteristics

Oxygen. Dissolved oxygen, which is essential to all aerobic aquatic organisms and strongly affects the solubility of various chemicals in the aquatic environment (Hutchinson 1957, 1967; Wetzel 1975), is an important water quality parameter. Because aquatic plants, through photosynthesis, respiration, and decomposition affect oxygen levels, there have been a large number of studies (Purdy 1916; Thienemann 1928; Rudolfs and Huekelekian 1931; Juday and Birge 1932; Olson 1932; Tomlinson 1935; Eberly 1959, 1963, 1964; Wetzel 1966a, 1966b; and many others) which have documented how aquatic plants affect oxygen content of aquatic systems. Purdy (1916) showed that aquatic plants, through photosynthesis, caused oxygen concentrations to increase during the day, but through respiration caused nocturnal reductions. This diel fluctuation in oxygen concentration (Figure 14) has also been described for other systems (Olson 1932; Welch 1952; Hutchinson 1957; Wetzel 1975) and seems to be a general phenomenon of aquatic systems.

Verduin (1956) summarized the literature on aquatic plant primary production in lakes and suggested that aquatic plants generally contribute 42-57 lb of oxygen per acre per day. This addition of oxygen to the aquatic environment has been shown to significantly affect distribution and percentage saturation of oxygen in lakes. Woodbury (1941) reported that dense algal blooms could produce supersaturation in open water of lakes. Juday and Birge (1932), Eberly (1959, 1963, 1964), and Wetzel (1966a, 1966b) have shown that algal growth in the region of the thermocline in some lakes can cause increases in oxygen concentration and may result in supersaturation at the thermocline. Wetzel (1975) has shown that vertical stratification of oxygen can occur in luxuriant stands of submersed macrophytes (Figure 14). Increases in oxygen levels were found near the surface of the macrophyte beds.

Although aquatic plants can increase the oxygen content of water and generally produce an excess of oxygen through photosynthesis, plants can also decrease oxygen concentration both directly and indirectly. Plant respiration does not normally reduce oxygen to amounts critical for survival of aquatic animals, but intense respiration coupled with high water temperature may stress or kill animals (Olson 1932; Prescott 1939). The major mechanism by which aquatic plants contribute to reduction of oxygen in the aquatic environment is through decomposition (Tomlinson 1935; Sears 1936; Hutchinson 1936; Moore 1942; Thomas 1960; Ruttner 1963; Wetzel 1975). In highly productive aquatic systems, plant decomposition can cause anoxia in the hypolimnion. Even in unstratified lakes, oxygen concentration may be reduced and fish killed after the death of large quantities of aquatic plants. Aquatic plants, however, do not necessarily have to die to reduce oxygen levels. Floating, floating-leafed and submergent aquatic macrophytes, by reducing wave action and water circulation, can prevent physical reaeration of the aquatic environment. Lynch et al. (1947), Ultsch (1973), and Wetzel (1975) have provided data that indicate that oxygen levels under mats of water hyacinth, alligator-weed, and Elodea can be reduced to near anoxia (Figure 14; Table 4).

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Table 4.
Comparisons of pH, temperature, dissolved O_2 and dissolved CO_2 in hyacinth and "open water" areas. March - October is the growing season of *Eichhornia* in Gainesville, Florida. From Ultsch, 1973.

	Hyacinths			"Open" water		
	Yearly	Nov.-Feb.	March-Oct.	Yearly	Nov.-Feb.	March-Oct.
Top Temp. (°C)	21.5	13.5	25.5	21.5	13.0	25.5
Bottom Temp. (°C)	19.0	12.0	23.0	20.0	13.0	23.5
Top pH	5.4	5.7	5.2	5.6	5.9	5.5
Bottom pH	5.3	5.6	5.1	5.5	5.8	5.4
Top O_2 (ppm)	4.2	5.6	3.5	6.4	8.1	5.6
Bottom O_2 (ppm)	1.2	2.5	0.6	4.5	7.2	3.2
Top CO_2 (ppm)	31	16	39	13	7	16
Bottom CO_2 (ppm)	51	26	63	29	8	40

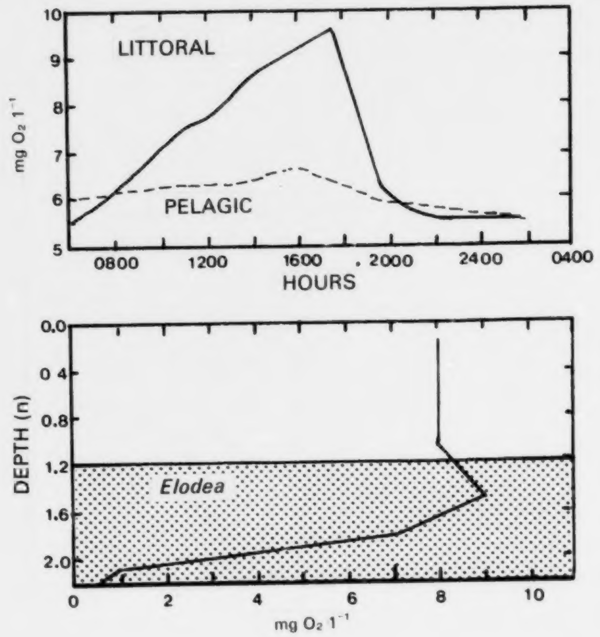

Figure 14. Upper: Changes in dissolved oxygen in the littoral and open water areas over a diel period in eutrophic Winona Lake, Indiana, 9 August 1922. (From data of Scott, 1924). Lower: Vertical stratification of oxygen within the littoral zone of Parvin Lake, Colorado, 9 July 1955, in a luxuriant stand of the submersed macrophyte Elodea. (Generated from data of Buscemi, 1958). From Wetzel, 1975.

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Despite the large amount of data now available on the relationship of aquatic plants to oxygen concentration in aquatic systems, there is a general lack of information on aquatic plant biomass and observed oxygen changes. For example, it has long been recognized that oxygen depletion in the hypolimnion of productive lakes occurs. There is, however, no reliable method to predict the quantity of aquatic plants needed by aquatic systems to maintain satisfactory oxygen regimens. Nor is there a reliable method to predict oxygen depletion in unstratified lakes, although Barica (1975) has provided data which might make it possible to predict summer fish kills (Figure 15). Water resource managers need quantitative information that can be used to predict the relationship of aquatic plant biomass to oxygen if oxygen depletions are to be prevented. Research should be conducted on a large number of lakes, such as that of Barica (1975), in order to determine general patterns. This type of information could provide water resource managers with predictive equations.

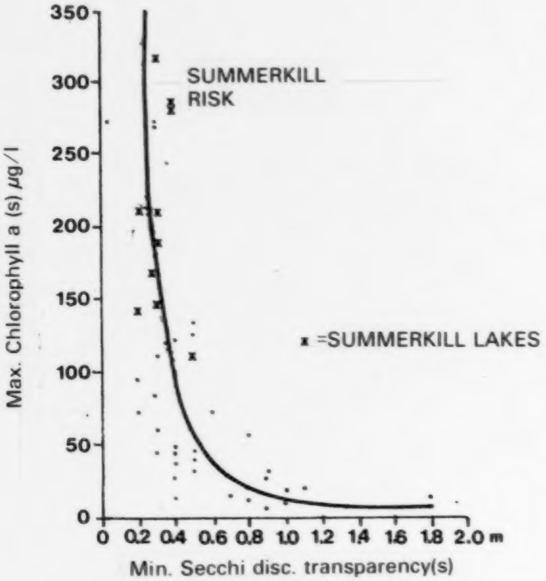

Figure 15. Relationship between the maximum chlorophyll a and minimum Secchi disc transparencies in various prairie ponds (Erickson-Elphinstone district of southwestern Manitoba). From Barica, 1975.

Carbon Dioxide, pH, Alkalinity. Most natural waters are buffered principally by a carbon dioxide - bicarbonate system. As CO_2 dissolves in water it forms carbonic acid: $CO_2 + H_2O \rightarrow H_2CO_3$ (2). Because carbonic acid is a weak acid, it dissociates rapidly:

$H_2CO_3 \rightarrow H^+ + HCO_3^-$ (3); $HCO_3^- \rightarrow H^+ + CO_3$ (4).

After equilibrium is established the bicarbonate and carbonate ions dissociate:

$HCO_3^- + H_2O \rightarrow H_2CO_3 + OH^-$ (5);
$CO_3^= + H_2O \rightarrow HCO_3^- \rightarrow HCO_3^- = OH^-$ (6);
$H_2CO_3 \rightarrow H_2O = CO_2$ (7).

The total quantity of carbon dioxide that passes into natural water, however, is not determined only by the pressure and solubility of carbon dioxide, but by the formation of bicarbonates with alkaline earth metals. For example, carbonic acid effectively solubilizes calcium carbonate, forming calcium biocarboate:

$H_2CO_3 + CaCO_3 \rightarrow Ca(HCO_3)_2$ (8); which dissolves in water:
$Ca(HCO_3)_2 \rightarrow Ca_{++} + 2HCO_3^=$ (9).

These reactions influence not only the carbon dioxide content of water but also the pH and alkalinity of natural waters.

Because aquatic plants utilize carbon dioxide during photosynthesis and release it during respiration, they strongly influence carbon dioxide content pH, and alkalinity of water. In well-buffered natural waters, the effect of aquatic plants may be minimal, but in poorly buffered systems aquatic plants may have a major impact. For example, Oosting (1933) showed that aquatic plants could deplete carbon dioxide in the upper water levels of Snail Lake (Table 5) while plant respiration or decomposition maintained higher carbon dioxide concentration at depth. Uptake and release of carbon dioxide can also produce diel variations in carbon dioxide, pH, and alkalinity (Figure 16), as noted by Unni (1972) and Van et al. (1976), and cause vertical variations. Oosting (1933) observed sharp pH gradients within aquatic macrophyte beds. At the surface of the beds he recorded a pH of 8.9 while only 2.5 feet below the leaves a significantly lower pH of 6.9 was found. Wetzel (1975) showed that similar vertical stratifications (Figure 17) in pH could occur also in the open water areas of hypereutrophic Wintergreen Lake.

Many aquatic organisms can withstand large variations in pH: thus, changes caused by aquatic plants seldom kill organisms. However, some organisms are extremely sensitive to pH changes and those caused by aquatic plants could affect these organisms. Carbon dioxide concentration, pH, and alkalinity, however, affect many of man's water uses, and it is extremely important to understand how aquatic plants affect carbon dioxide concentration, pH, and alkalinity. Although the chemical reactions are well understood and there is ample information on changes aquatic plants may cause, water resource managers have a difficult time predicting chages in carbon dioxide concentration, pH, and alkalinity that might result from aquatic plant activity. Quantitative data relating changes in carbon dioxide, pH, and alkalinity to aquatic plant biomass in aquatic systems are lacking as most studies have been qualitative. Before water resource managers can predict the effect of aquatic plant biomass on carbon dioxide, pH, and alkalinity in different aquatic systems, data will have to be obtained from a large number of aquatic systems.

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Table 5.
Carbon dioxide: vertical variation 1928.
From Oosting, 1933.

	Snail Lake (cc./L)			
	Aug. 4	Aug. 11	Aug. 18	Aug. 25
6 in.	.25	0	0	0
3 ft.	.51	0	0	0
6 ft.	.51	0	0	0
9 ft.	.51	0	0	0
12 ft.	.51	0	0	0
15 ft.	.51	1.0	0	0
18 ft.	.76	1.5	.51	.3
21 ft.	.76	1.8	1.00	.5
22 ft.	1.00			
24 ft.		2.0	1.80	3.0
26 ft.		2.5	2.30	3.3

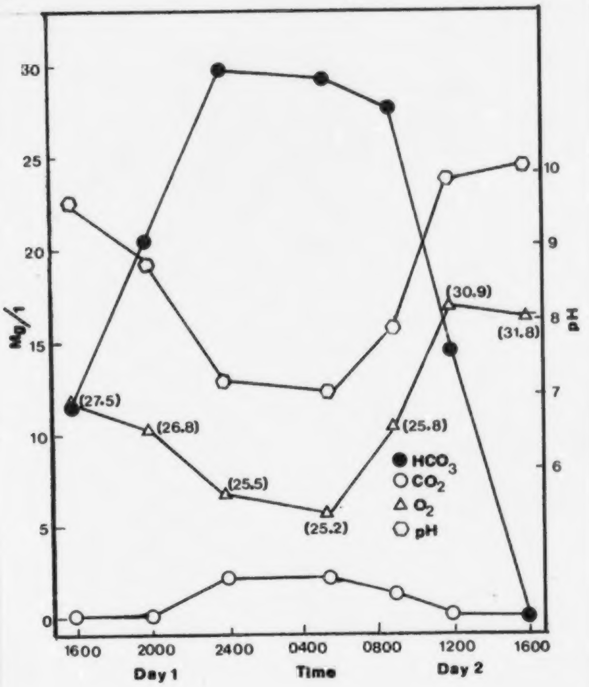

Figure 16. Diurnal fluctuations in free CO_2, HCO_3^-, O_2, and pH measured in the surface 5 cm of water over a *Hydrilla verticillata* mat. Data were collected on October 14 and 15, 1975, at Lake Killarney, Florida. Figures in parentheses refer to the water temperature (C) at the time of sampling. From Van et al., 1976.

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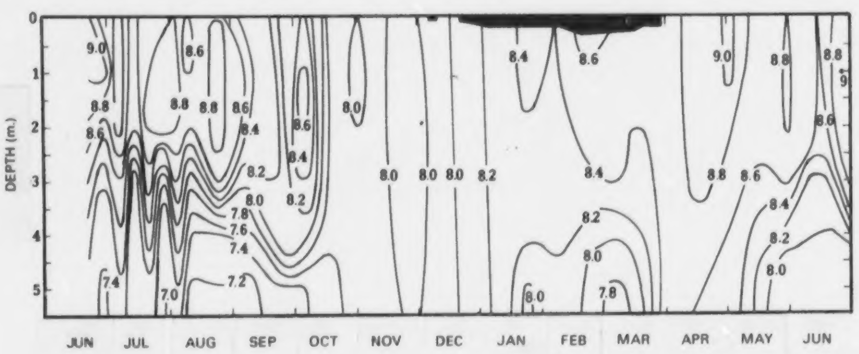

Figure 17. Depth-time diagram of isopleths of pH in hypereutrophic Wintergreen Lake, Michigan, 1971 - 1972. Opaque area = ice cover to scale. From Wetzel, 1975.

Water Salinity. Salinity of freshwaters is usually dominated by the cations of magnesium, sodium, potassium, and calcium and the anions of chloride, sulfate, and carbonate. Concentrations of magnesium, sodium, potassium, and chloride are relatively conservative and undergo minor spatial and temporal fluctuations due to aquatic plant utilization (Figure 18; Wetzel 1975). Surface water reductions in potassium (Figure 18) may, however, be due to aquatic plant uptake and subsequent aquatic plant sedimentation (Barrett 1957; Wetzel 1975). Plant decomposition has also been suspected of occasionally releasing large amounts of magnesium to the aquatic environment (Wetzel 1975).

Aquatic plants, however, significantly affect temporal and spatial distribution of calcium and carbonate concentrations. For example, calcium (Figure 19) and carbonate concentrations in Wintergreen Lake, Michigan, underwent marked seasonal fluctuations (Wetzel 1975) that are typical of many hardwater lakes (Welch 1952; Hutchinson 1957; Wetzel 1975). Such fluctuations result when aquatic plants utilize carbon dioxide during photosynthesis. Uptake of carbon dioxide affects the chemical equilibrium of the waters by: $Ca^{++} + 2HCO_3^- \rightarrow CO_2 + H_2O + CaCO_3$. This process results in precipitation of calcium carbonate ($CaCO_3$). Otsuki and Wetzel (1974) observed that aquatic plant photosynthesis, by inducing precipitation of calcium carbonate, was responsible for decalcification of surface waters. The loss of calcium also reduced the specific conductance of the surface waters. Because some calcium carbonate can be resolubilized by carbon dioxide in deeper waters, precipitation of calcium carbonate from surface waters can cause an increase in concentration of calcium and carbonate ions in deeper waters.

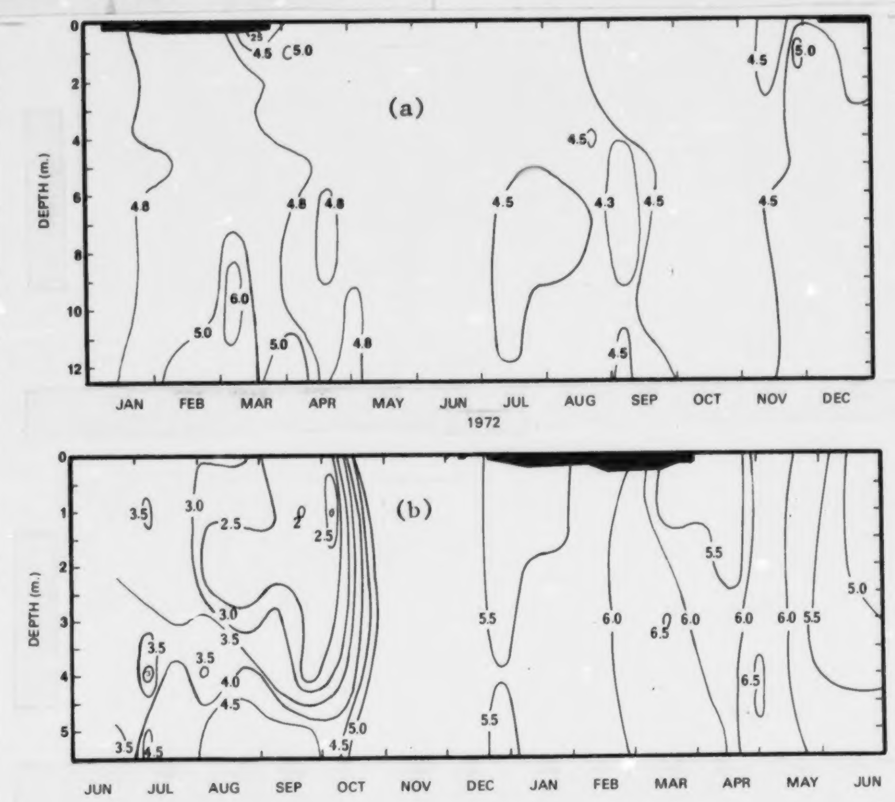

Figure 18. Depth-time distribution of isopleths of sodium concentrations (mg Na^+ 1^{-1}) of Lawrence Lake, 1972 (a), potassium concentrations (mg K^+ 1^{-1}) of hypereutrophic Wintergreen Lake, Michigan, 1971-1972 (b), chloride concentrations (mg Cl^- 1^{-1}) of eutrophic Little Crooked Lake, Whitley County, Indiana, 1964 (c), and magnesium concentrations (mg Mg^{++} 1^{-1}) of hardwater Lawrence Lake, Michigan, 1972 (d). Opaque areas = ice-cover to scale. From Wetzel, 1975.

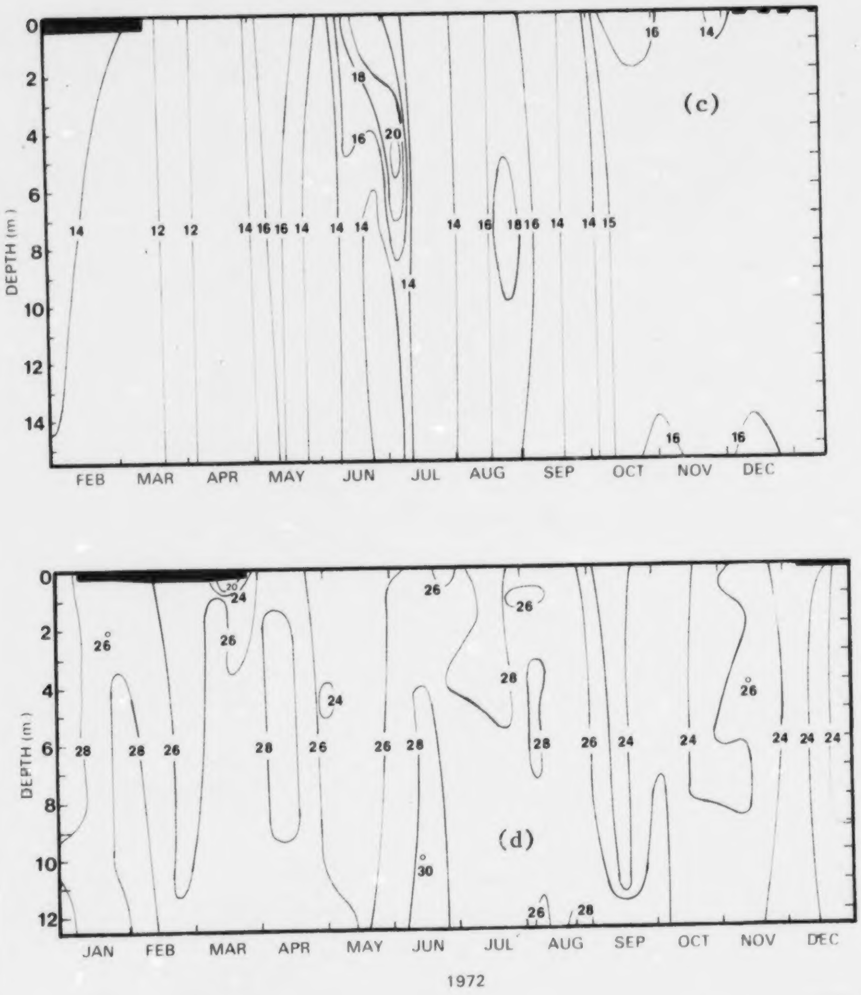

Figure 18. Continued.

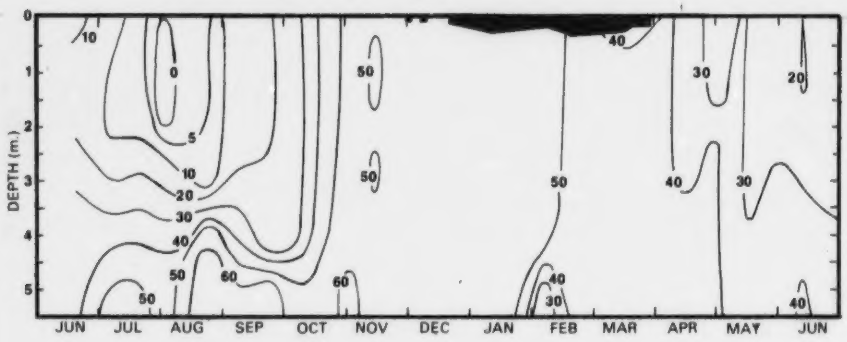

Figure 19. Depth-time distribution of isopleths of calcium concentrations (mg Ca^{++} l^{-1}) of hypereutrophic Wintergreen Lake, Kalamazoo County, Michigan, 1971 - 72. Opaque area = ice cover to scale. From Wetzel, 1975.

Past research has documented the possible impacts of aquatic plants on water salinity. This research has, however, been basically descriptive. There is a lack of quantitative information on the relationship of aquatic plant biomass to salinity. For example, there are no data to predict the effect of aquatic plant biomass on the quantity of calcium carbonate precipitated. Future research should relate changes in water salinity to the quantity of plants present.

Nitrogen. Aquatic plants, through utilization of nitrogen compounds during growth, can affect temporal and spatial distribution of nitrogen. For example, Stadelmann (1971) observed that uptake of ammonia and nitrate reduced the concentration of these compounds to very low levels in lake surface water. Plant decomposition in the bottom water, however, caused an increase in ammonia concentration (Figure 20). These changes are only a general pattern, as seasonal changes in nitrogen can very greatly among aquatic systems of different productivities. For a detailed discussion of general seasonal patterns in the transition from nutrient poor to nutrient rich aquatic systems, consult Hutchinson (1957) or Wetzel (1975).

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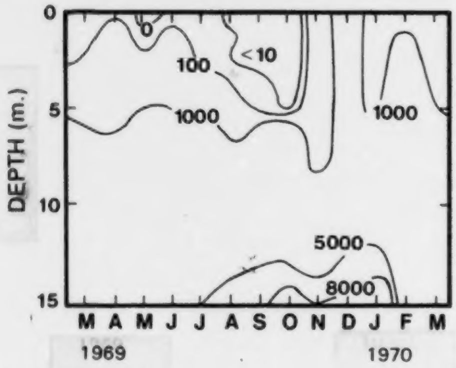

Figure 20. Depth-time diagram of the concentrations of ammonia (mg NH_3-N l^{-1}), Rotsee, Switzerland, 1969-70. (Redrawn from Stadelmann, 1971). From Wetzel, 1975.

Blue-green algae, because of their ability to fix atmospheric nitrogen (Burris et al. 1943; Horne and Fogg 1970; Rusness and Burris 1970; Horne et al. 1972), significantly affect nitrogen budgets of aquatic systems and thus affect their productivity. Wetzel (1975) noted that nitrogen fixation in waters without blue-green algae is insignificant, but as blue-green algal populations increase, nitrogen fixation becomes significant. If fixation of nitrogen becomes great enough, algal blooms may occur in waters that were otherwise nitrogen limited. Within a given water body, Rusness and Burris (1970) noted diel variations in nitrogen fixation with fixation being greatest at midday and lowest at night (Figure 21). Wetzel (1975), using data from Horne and Fogg (1970), showed that nitrogen fixation may vary with depth and that the pattern of fixation varies among lakes (Figure 22). Billaud (1968), Horne and Fogg (1970), and Toetz (1973) noted that nitrogen fixation rates decline abruptly with decreases in blue-green algal populations and are greatly reduced during the winter in northern lakes.

Although research has documented the effects of aquatic plants on the nitrogen cycle of the aquatic environment, there is still a lack of quantitative data. Before water resource managers will be able to predict the total impact of aquatic plants on the nitrogen cycle of aquatic systems, relationships now known will have to be related to aquatic plant biomass. Because nitrogen is often believed to be a limiting plant nutrient, additional research is needed to determine if nitrogen fixation by blue-green algae supplies a quantitatively important part of the nitrogen budget of aquatic systems. This information is especially important to water resource managers who are charged with reducing nutrient inputs to aquatic systems in order to control aquatic plant growth. If blue-green algae can fix sufficient atmospheric nitrogen to support this growth and eventually the growth of other aquatic plants, reduction of nitrogen inputs resulting from human activities will have little impact on aquatic plant growth.

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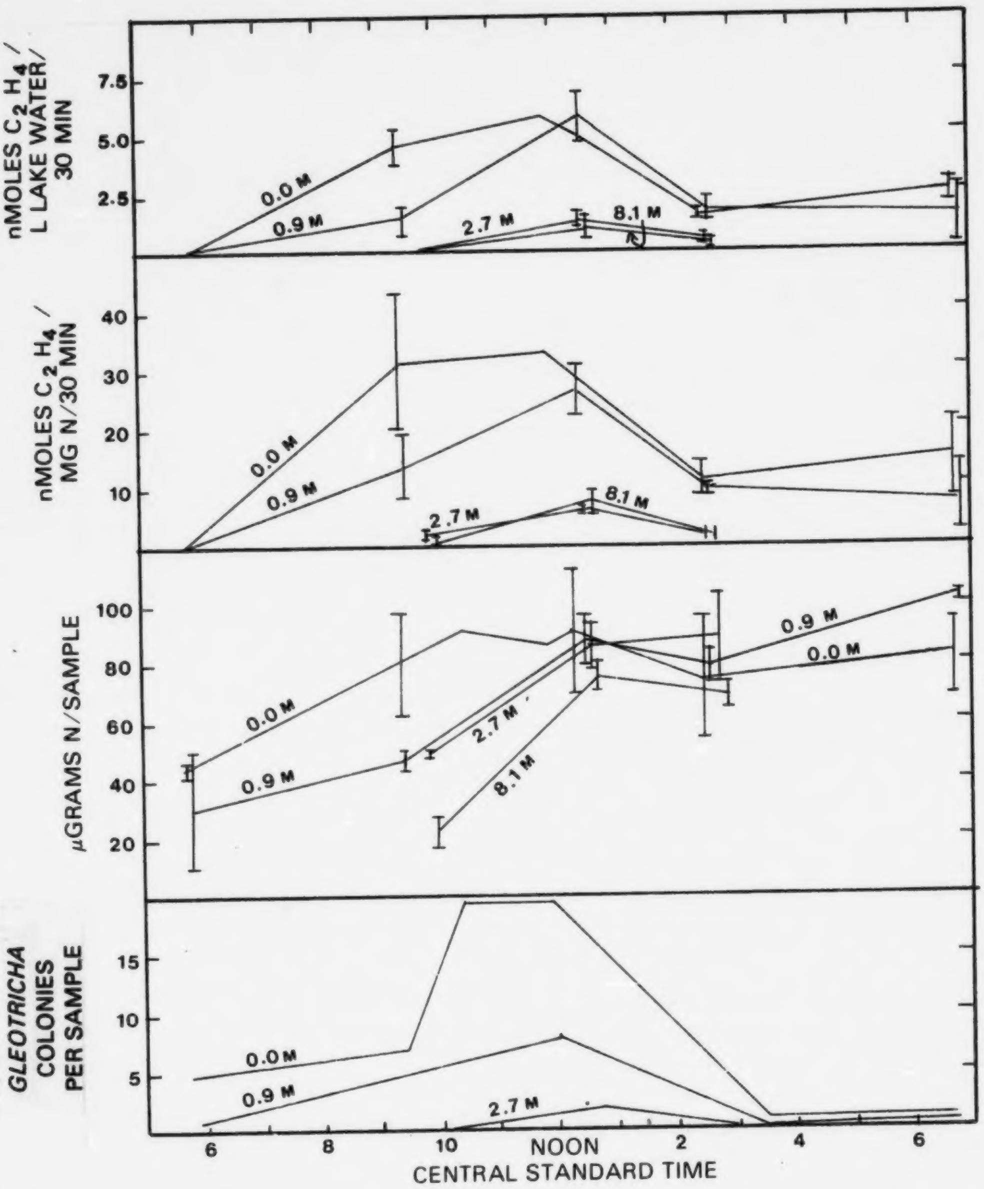

Figure 21. Comparison of in situ acetylene reduction throughout the day of 21 August 1968 by samples from various depths in Lake Mendota. The ranges between replicates are indicated, and the mean values are plotted. Numbers within brackets represent the depth in meters at which the samples were collected. Samples were incubated at the surface in a pail of water. From Rusness and Burris, 1970.

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Phosphorus. Studies on natural lakes (Sakamoto 1966; Dillon and Rigler 1974b; Jones and Bachmann 1976; Canfield 1979; and many others) have shown a strong correlation between total phosphorus concentrations and algal biomass (Figures 1 and 2), thus suggesting phosphorus is the element limiting algal biomass. In fact, phosphorus may be the element limiting all aquatic plant growth. For this reason, many studies have examined phosphorus dynamics in lakes.

Studies with radioactive phosphorus (Hutchinson and Bowen 1947, 1950; Coffin et al. 1948; Hayes et al. 1952; Hayes and Phillips 1958) have shown that aquatic macrophytes, attached algae, and phytoplankton rapidly assimilate phosphorus added to lake water. These studies along with studies by Solski (1962) and Nichols and Keeney (1973) have shown that upon death and decay the plants release a large portion of the phosphorus to the water. Recent studies (McRoy and Barsdate 1970; Boyd 1971; Reimold 1972; McRoy et al. 1972; Lie 1978) have suggested that rooted aquatic macrophytes can also assimilate phosphorus from the bottom sediments. This phosphorus can also be released to overlying waters. Lie (1978) calculated that aquatic plants recycled about 5000 kg of bottom sediment phosphorus per year in Lake Shagawa, Minnesota. This phosphorus release, according to Lie (1978), was sufficient to maintain the high phosphorus concentrations observed in Lake Shagawa after nutrient reduction programs were initiated.

Aquatic plants can also indirectly affect cycling of phosphorus. When large quantities of organic matter are sedimented to bottom water, decomposition often produces anoxia. Mortimer (1941, 1942, 1971) has shown that when the oxygen supply is depleted, phosphorus is released from sediments. Mortimer further noted that this release increased markedly as the redox potential decreased.

There is little doubt that aquatic plants can affect phosphorus concentrations in lakes. Perhaps the most pressing research needs involve elucidation of the role of aquatic macrophytes in the cycling of phosphorus. Laboratory studies strongly indicate aquatic macrophytes assimilate phosphorus in water from bottom sediments. These laboratory studies also indicate that macrophytes release phosphorus. In order to manage algal problems, water resource managers need to have quantitative information on the quantity of inflowing phosphorus that aquatic macrophytes will assimilate and how these quantities vary with changes in aquatic plant biomass. Quantitative information must also be obtained on how much phosphorus aquatic macrophytes release to water if nutrient reduction programs are to be effective. Lie's (1978) work suggested that aquatic macrophytes release sufficient phosphorus to maintain Lake Shagawa in a eutrophic state. An important question is how much reduction in aquatic plant biomass is required to reduce phosphorus release to tolerable rates. Additional research should also be conducted to determine how different quantities of aquatic macrophytes affect the phosphorus budgets of different aquatic systems.

Silica, Iron, Manganese and Sulfur. Seasonal variations in silica concentrations (Figure 23) are directly correlated with development of diatom populations (Meloche et al. 1938; Lund 1964; Wetzel 1975). Kilham (1971) has observed that diatoms can reduce surface water silica concentrations to less than a few micrograms per liter. After death and sedimentation, mineralization causes increases in hypolimnetic silica concentrations (Stangenburg 1961).

Aquatic plants seldom directly influence the cycling of iron, manganese, and sulfur in the aquatic environment, although aquatic plants do have metabolic demands for these elements. The major effect of aquatic plants on cycling of these elements results from depletion of oxygen during decomposition of aquatic plant remains. With loss of oxygen from water overlying sediment, redox potential falls (Wetzel 1975), and when low enough, iron and manganese can be released from the sediments. Sulfate can be reduced to hydrogen sulfide and additional hydrogen sulfide can be produced by bacterial decomposition of sulfur-containing organic matter.

There is a lack of quantitative data that can be used to predict the effect of aquatic plant biomass on the silica, iron, manganese, and sulfur cycles of aquatic systems. Because these elements are important water quality parameters in many domestic and industrial water uses, water resource managers need data that can be used to predict their concentrations. It is especially important to be able to relate anoxia to amount of dead organic matter in order to predict release of iron, manganese, and sulfur from the sediments.

Taste, Odor, and Toxic Substances. Taste and odor are major water quality problems. Excessive growth of certain aquatic plants can produce taste and odor in water (Palmer 1959; Neel et al. 1961). The plants that cause taste and odor problems most often are phytoplankton, especially certain diatoms, blue-green algae, and pigmented flagellates (Palmer 1959). These algae can impart a fish taste to the water as well as cause a fishy, musty, septic, or pigpen odor (Table 6).

Studies by Prescott (1948), Ingram and Prescott (1954), and Rose (1953, 1954) suggest that blue-green algae and dinoflagellates can produce toxic substances that can kill fish, birds, and mammals. These toxic substances seem to be more lethal when there are dense growths of algae. Prescott (1968) noted that domestic animals such as horses, cattle, sheep, hogs, and birds have been killed or made severely ill by drinking algae-infested water. He noted that death often resulted within 1 to 24 hours. Human consumption of fish and shellfish which have injested large quantities of dinoflagellates, especially *Gonyaulax*, can also result in death or serious poisoning (Prescott 1968).

A major research need is definition of factors that enhance growth of algae that produce taste, odor, and toxic substances. In addition, quantitative information is needed on the amount of aquatic plant biomass that produces problems. With this information it might then become possible to manage taste and odor prolems as well as toxicity problems.

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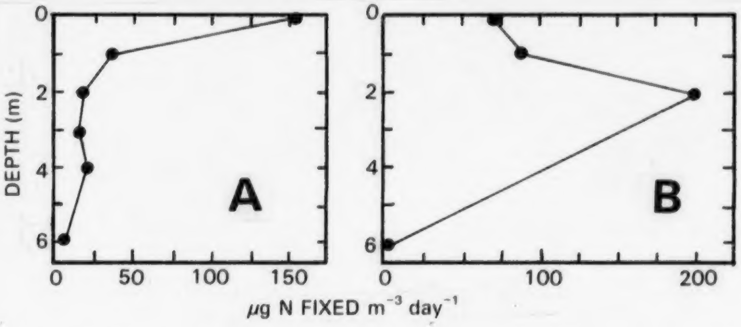

Figure 22. Variations in nitrogen fixation with depth (A) in Lake Windermere, and (B) in Esthwaite Water, England, 30 August 1966. (Modified from Horne and Fogg, 1970). From Wetzel, 1975.

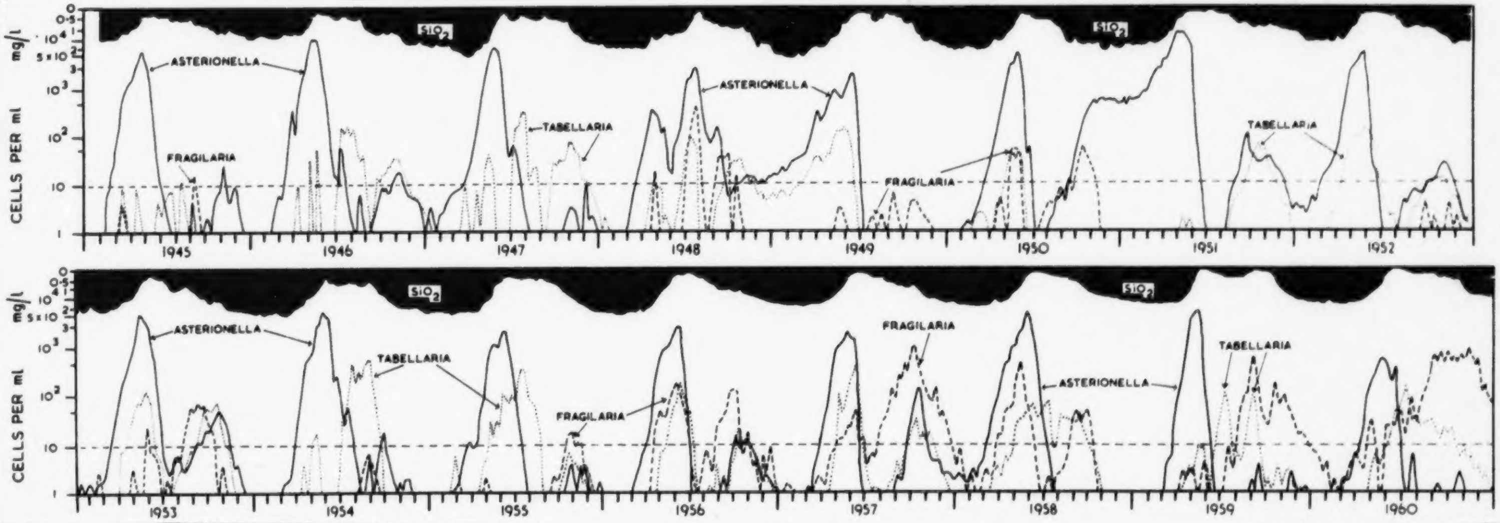

Figure 23. The periodicity of the diatom algae Asterionella formosa, Fragilaria crotonensis, and Tabellaria flocculosa in relation to fluctuations in the concentration of dissolved silica, 0-5 m in Lake Windermere, England, 1945 - 1960. From Lund, 1964.

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Table 6.
Odors, tastes, and tongue sensations associated with algae in water.
From The Practice of Water Pollution Biology, 1969.

Algal genus	Odor when algae are —		Taste	Tongue sensation
	Moderate	Abundant		
Actinastrum		Grassy, musty		
Anabaena	Grassy, nasturtium, musty.	Septic		
Anabaenopsis		Grassy		
Anacystis	Grassy	Septic	Sweet	
Aphanizomenon	Grassy, nasturtium, musty.	Septic	Sweet	Dry.
Asterionella	Geranium, spicy	Fishy		
Ceratium	Fishy	Septic	Bitter	
Chara	Skunk, garlic	Spoiled, garlic		
Chlamydomonas	Musty, grassy	Fishy, septic	Sweet	Slick.
Chlorella		Musty		
Chrysosphaerella		Fishy		
Cladophora		Septic		
(Clathrocystis)				
Closterium		Grassy		
(Coelosphaerium)				
Cosmarium		Grassy		
Cryptomonas	Violet	Violet	Sweet	
Cyclotella	Geranium	Fishy		
Cylindrospermum	Grassy	Septic		
Diatoma		Aromatic		
Dictyosphaerium	Grassy, nasturtium	Fishy		
Dinobryon	Violet	Fishy		Slick.
Eudorina		Fishy		
Euglena		Fishy	Sweet	
Fragilaria	Geranium	Musty		
Glenodinium		Fishy		Slick.
(Gloeocapsa)				
Gloeocystis		Septic		
Gleotrichia		Grassy		
Gomphosphaeria	Grassy	Grassy	Sweet	
Gonium		Fishy		
Hydrodictyon		Septic		
Mallomonas	Violet	Fishy		
Melosira	Geranium	Musty		Slick.
Meridion		Spicy		
(Microcystis)				
Nitella	Grassy	Grassy, septic	Bitter	
Nostoc	Musty	Septic		
Oscillatoria	Grassy	Musty, spicy		
Pandorina		Fishy		
Pediastrum		Grassy		
Peridinium	Cucumber	Fishy		
Pleurosigma		Fishy		
Rivularia	Grassy	Musty		
Scenedesmus		Grassy		
Spirogyra		Grassy		
Staurastrum		Grassy		
Stephanodiscus	Geranium	Fishy		Slick.
Synedra	Grassy	Musty		Slick.
Synura	Cucumber, muskmelon, spicy.	Fishy	Bitter	Dry, metallic, slick.
Tabellaria	Geranium	Fishy		
Tribonema		Fishy		
(Uroglena)				
Uroglenopsis	Cucumber	Fishy		Slick.
Ulothrix		Grassy		
Volvox	Fishy	Fishy		

Gleotrichia

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Impact of Aquatic Plants on Biological Characteristics

Productivity. In aquatic systems, plants are the primary producers of organic matter on which many organisms depend for food. Because most invesigators assumed phytoplankton were the primary producers of readily digeststable organic matter, there now exists extensive literature on algal primary production (Table 7; Wetzel 1975). Studies have shown that annual algal primary production rates can range from about 4 g $C/m^2/yr$ (Kalff and Welch 1974) to over 600 g $C/m^2/yr$ (Talling 1965; Lewis 1974) in natural lakes. Wetzel (1975) summarized the literature and noted that primary production rates in oligotrophic lakes ranged from 4 to 24 g $C/m^2/yr$ while mesotrophic lakes ranged from 75 to 250 g $C/m^2/yr$ and eutrophic lakes ranged from 350 to 700 g $C/m^2/yr$.

Although it has long been recognized that productivity is greater in nutrient-rich aquatic systems, there has been considerable debate on the factors that control algal primary productivity. In single lake studies, it has often been difficult to show that algal productivity increases with increases in nutrients. Smith (1979), by using data from a large number of lakes, has shown that mean growing season photosynthesis expressed per unit volume of the euphotic zone is highly correlated to mean total phosphorus concentrations (Figure 24) and mean chlorophyll concentrations (Figure 25). These data are particularly useful to water resource managers as they permit, for the first time, the prediction of the impact of changing nutrient inputs on algal productivity and the impact of controlling algal populations on algal productivity. Because the relationships in Figures 24 and 25 are derived from a group of lakes representing a wide range of limnological conditions, the results from Smith's (1979) study should be applicable to a range of lakes.

Although most of the work on aquatic plant production in the aquatic environment has centered on algal primary production, the early works of Rickett (1922, 1924) and Wilson (1935, 1937) showed that the weight of the total crop of aquatic macrophytes in lakes could be significant. Since this work, a number of studies have measured aquatic macrophyte biomass and productivity (Table 7). The importance of aquatic macrophytes to the total productivity of the aquatic environment, however, was often overlooked because many workers considered aquatic macrophytes to be an unimportant food source. Odum and de la Cruz (1963), however, suggested that the organic matter produced by aquatic macrophytes was an important source of detritus which supported aquatic organisms. Studies by Nygaard (1958), Westlake (1963), and Wetzel (1964) further suggested that aquatic macrophytes were extremely important in determining the total productivity of the aquatic environment. Recent studies have provided data which support this hypothesis (Table 8). Rich et al. (1971) showed that in a southern Michigan marl lake aquatic macrophytes contributed 48% of the total annual production while phytoplankton contributed 30%. Wetzel et al. (1972) showed that macrophytes contributed 51% of the total annual production in Lawrence Lake, Michigan, while phytoplankton contributed 25% and attached algae contributed 24%. These data strongly suggest that while photoplankton may contribute as much as 99% of the total production in some lakes (Schindler et al. 1973), the role of macrophytes and attached algae in other lakes must be considered.

Table 7.

Comparison of rates of primary production of phytoplankton in selected fresh waters of varying fertility[a] and representative estimations of annual above-ground biomass and productivity of aquatic macrophytes. From Wetzel, 1975.

Lake	Remarks	Mean Daily Productivity for Entire Year (mg C m^{-2} day^{-1})	Range Observed (mg C m^{-2} day^{-1})	Annual Productivity (g C m^{-2} $year^{-1}$)
OLIGOTROPHIC:				
Char, N.W.T., Canadian arctic (lat. 74°) (Kalff and Welch, 1974)	80% of total production by benthic flora	1.1	0-35	4.1
Meretta, N.W.T., Canada (Kalff and Welch, 1974)	Polluted by sewage	3.1	0-170	11
Castle, Calif. (Goldman, pers. comm.)	Deep, alpine	98	6-317	36
Lunzer Untersee, Austria (Jonasson, 1972)	Small, alpine	(123)	-	45
Lawrence, Mich. (Wetzel; cf. Table 14-7)	Small, hardwater; 7-year averge	112.6	5-497	41.1
Ransaren, Sweden (Rodhe, 1958)	Small lapplandic	—	23-66	—
Lake Superior, USA-Canada (Putnam and Olson, 1961)	Most unproductive of Laurentian Great Lakes	—	50-260	—
Lake Huron, USA-Canada (Vollenweider, et al., 1974)	Offshore stations	—	150-700	ca 100
Borax, Calif. (Wetzel, 1964)	Large, shallow saline lake (25% of total productivity)	249	10-524	91
Lake Michigan, USA (Vollenweider, et al., 1974)	Offshore stations	—	70-1030	ca 130
Lake Ontario, USA-Canada (Vollenweider, et al., 1974)	Offshore stations	—	60-1400	ca 180
MESOTROPHIC:				
Erken, Sweden (Rodhe, 1958)	Large, deep, naturally productive	285	40-2205	104
Clear, Calif. (Goldman and Wetzel, 1963)	Very large, shallow	438	2-2440	160
Esrom, Denmark (Jonasson and Mathiesen, 1959)	Large, moderately deep	370	23-422	260
Fureso, Denmark (Jonasson and Mathiesen, 1959)	Large, deep, many macrophytes	462	0-1380	168
Walter, Ind. (Wetzel, 1973)	Series of 4 interconnected marl lakes			
Basin A		418	102-1395	153
Basin B		210	12-535	77
Basin C		276	30-1048	101
Basin D		437	98-1458	160
Oliver, Ind. (Wetzel, 1973)	Large, deep, marl lake	336	32-775	123
Olin, Ind. (Wetzel, 1973)	Large, deep, marl lake	374	89-996	137
Martin, Ind. (Wetzel, 1973)	Deep, stained marl lake	561	27-1708	205
Pretty, Ind. (Wetzel, 1966b)	Moderate-sized, deep, marl lake			
1963		440	68-1850	161
1964		305	6-895	111

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Lake	Remarks	Mean Daily Productivity for Entire Year (mg C m $^{-2}$ day $^{-1}$)	Range Observed (mg C m $^{-2}$ day $^{-1}$)	Annual Productivity (g C m $^{-2}$ year $^{-1}$)
Crooked, Ind. (Wetzel, 1966b)	Large, deep, hardwater lake			
1963		469	142-1364	171
1964		359	23-870	131
Little Crooked, Ind. (Wetzel, 1966b)	Small, deep, kettle lake			
1963		618	263-1903	226
1964		508	9-2431	218
Goose, Ind. (Wetzel, 1966a)	Small kettle lake	729	166-1753	266
Lake Erie, USA-Canada (Vollenweider, et al., 1974)				
Western stations		—	30-4760	(310)
Central stations		—	120-1690	(210)
Eastern stations		—	140-1440	(160)
EUTROPHIC:				
Wintergreen, Mich. (Wetzel, et al., unpublished)	Shallow, extensive nutrient loading	1012	60-2240	369
Frederiksborg Slotsso Denmark (Nygaard, 1955)	shallow, enriched	1030	12-4160	376
Minnetonka, Minn. (Megard, 1972)	Extremely complex basin. large, deep	(820)	-	(300)
Sollerod So, Denmark (Steemann Nielsen, 1955)		1430	0-3800	522
Sylvan, Ind. (Wetzel, 1966a; cf. Table 14-9)	Complex basin, large, shallow	1564	9-4959	570
Lanao, Philippines (Lewis, 1974)	Large, deep tropical lake	1700	400-5000	620
Victoria, Africa (Talling, 1965)	Large, deep, equatorial lake	1750	1700-3800	640
DYSTROPHIC:				
Kattehale Mose, Denmark (Nygaard, 1955)	Very shallow, acidic, peat bog	80	0-400	29
Smit Hole, Ind. (Wetzel, 1973)	Shallow, humic stained	194	24-5960	71
Store Gribso, Denmark (Nygaard, 1955)	Deep, acidic, humic stained	230	4-680	84
Grane Langso, Denmark (Nygaard, 1955)	Deep, acidic, humic stained	248	20-880	91

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Type and Lake	Seasonal Maximum Biomass or Above-Ground Biomass (g dry m^{-2})	Productivity (g m^{-2} year^{-1})	Source
SUBMERGENTS DOMINATING:			
Trout L., Wisc. (softwater)	0.07	—	Wilson, 1941
Sweeney L., Wisc. (softwater)	1.73	—	Wilson, 1937
Weber L., Wisc. (softwater)	16.8	—	Potzger and Engel, 1942
Lowes L., Scotland (dystrophic)	32	—	Spence, et al., 1971
Spiggie L., Scotland (dystophic)	100	—	
L. Mendota, Wisc. (hardwater)	202	—	Rickett, 1921
Lawrence L., Mich. (hardwater)			
Submersed *Scirpus subterminalis*	338	565	Rick, et al., 1971
Chara	110	155	
Annuals	130	199	
Croispol, Scotland	400	—	Spence, et al., 1971
Borax L., Calif. (saline lake. *Ruppia*)	60	64	Wetzel, 1964
River Ivel, England (*Berula;* very fertile)	500	—	Edwards and Owens, 1960
River Test, England (*Ranunculus*)	100-400	—	Owens and Edwards, 1961
River Yare, England (*Potamogeton*)	380	—	Owens and Edwards, 1962
Saline channels, Puerto Rico (*Thalassia*)	700-7300	—	Burkholder, et al., 1959
FLOATING:			
New Orleans, La. (*Eichhorina*)	630-1472	1500-4400	Penfound and Earle, 1948
EMERGENTS DOMINATING:			
Ladoga L., USSR	0.4-10.7	—	Raspopov, 1971
Onega L., USSR	0.1-33	—	
Blanket bog, England (*Sphagnum*)			
on hummocks	—	180	Clymo and Reddaway, 1971, 1974
in pools	—	290	
on "lawns"	—	340	
Polish lakes (emergent species)	440-830	—	Szezepańska, 1973
Minnesota wetlands (*Carex*)	850	738	Bernard, 1973
Surlingham Broad, England (*Glyceria, Typha,* and *Phragmites*)	800-1100	—	Buttery and Lambert, 1965
Opatovicky Pond, Czechoslovakia (*Phragmites*)			
Aerial	1100-2200	—	Dykyjová and Hradecka, 1973
Below ground	6000-8560	—	
Cedar Creek, Minn. (*Typha*)	4640	2500	Bray, et al., 1959

✓ Rick, et al. 1971

Hradecka

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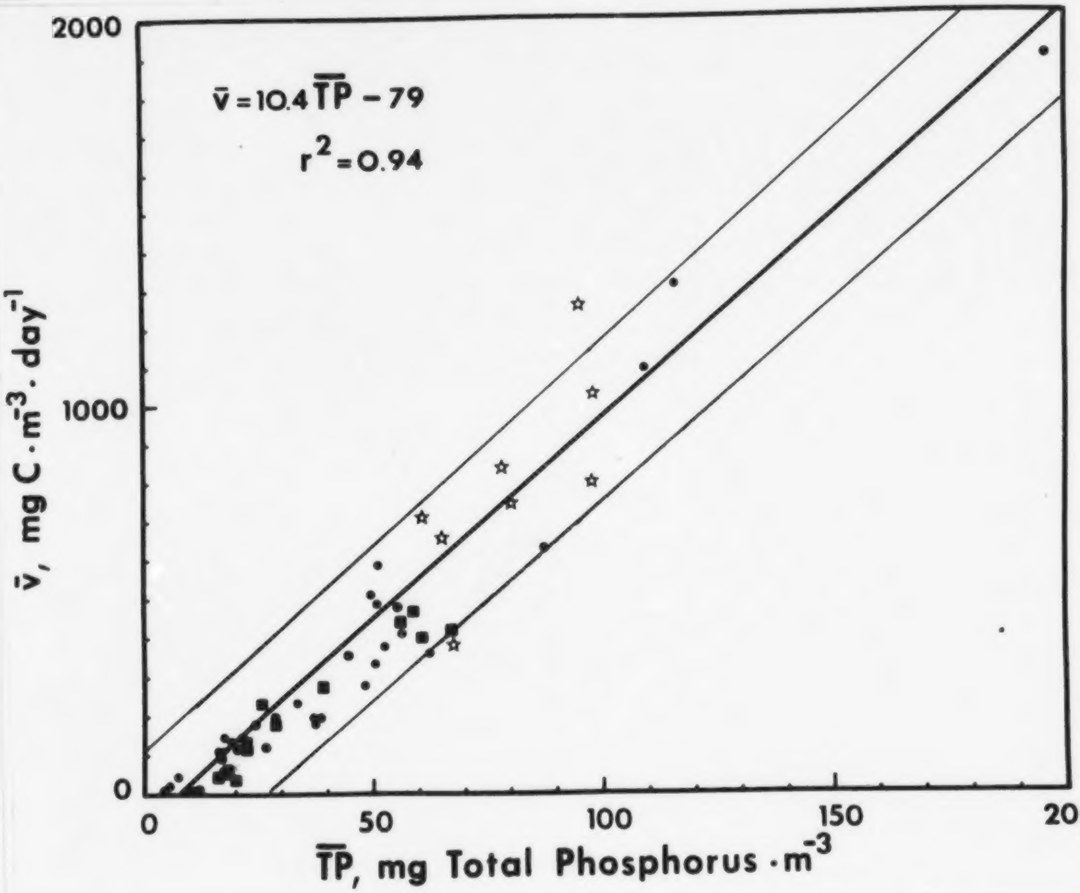

Figure 24. Correlation between growing season mean photosynthesis (per unit volume euphotic zone) and mean total phosphorus concentration for 38 north temperate lakes. From Smith, 1979.

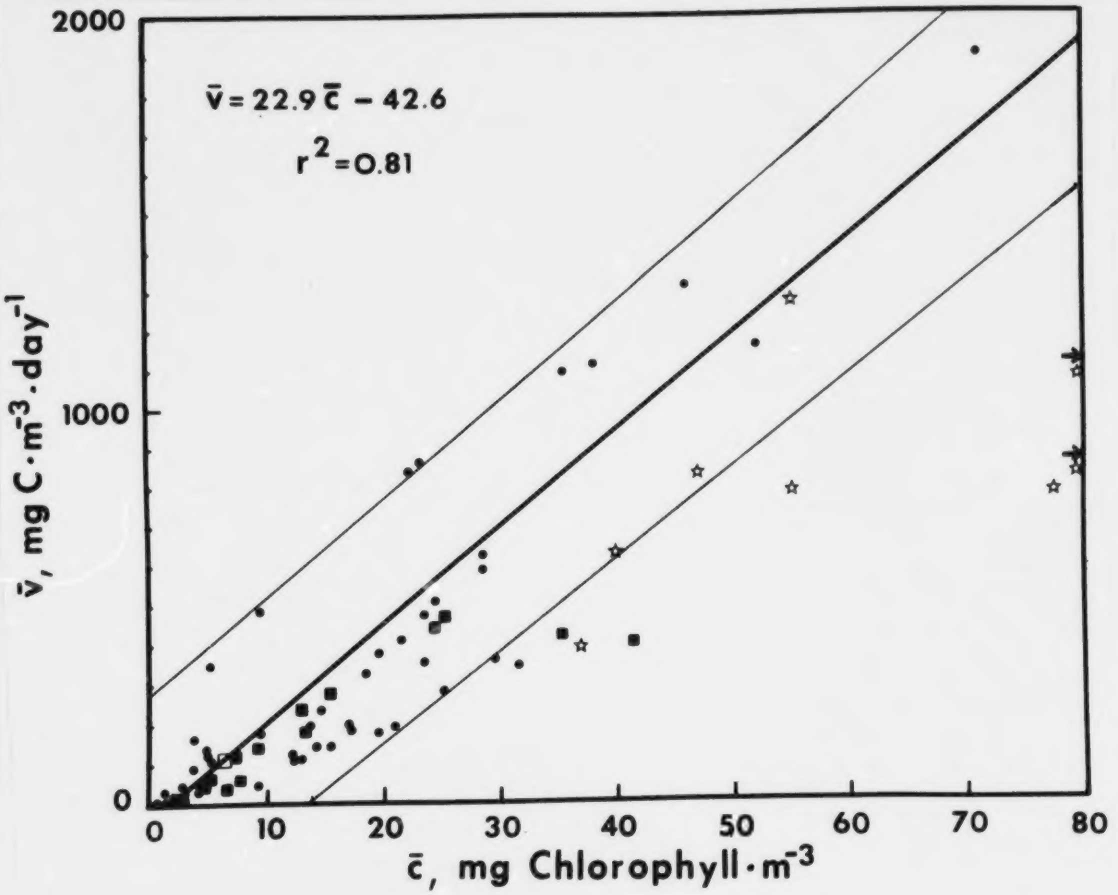

Figure 25. Correlation between growing season mean photosynthesis (per unit volume euphotic zone) and mean chlorophyll concentration for 49 North American lakes. Limits shown are 95% confidence limits for individual points around regression line. ■ - Lake Washington; □ - Tuttle Creek Reservoir; ☆ - European lakes. From Smith, 1979.

The recent work of Smith (1979) is most promising. Additional research on other lakes should be done to further expand the data base, but more important quantitative data should be gathered on aquatic macrophytes and organisms. Studies by Nygaard (1958), Westlake (1963) and Wetzel (1964) further suggested that aquatic macrophytes were extremely important in determining total productivity of the aquatic environment. Recent studies have provided data that support this hypothesis (Table 8). Rich et al. (1971) showed that in a southern Michigan marl lake aquatic macrophytes contributed 48% of the total annual production while phytoplankton contributed 30%. Wetzel et al. (1972) showed that macrophytes contributed 51% of the annual production in Lawrence Lake, Michigan, while phytoplankton contributed 25% and attached algae 24%. These data strongly suggest that, while phytoplankton may contribute as much as 99% of the total production in some lakes (Schindler et al. 1973), the role of macrophytes and attached algae must be considered in other lakes.

Table 8.
Examples of annual net productivity of phytoplankton, littoral algae, and macrophytes of several lakes in which productivity estimates of attached algae were made on natural substrata. From Wetzel, 1975.

Lake	Area (ha)	Mean Depth (m)	Annual Mean (mg C m^{-2} day^{-1})	Annual Mean (kg C $lake^{-1}$ day^{-1})	Kg C ha^{-1} of Lake Surface $year^{-1}$		Remarks
Borax, Calif.	39.8	>0.5				(%)	Saline lake; benthic algae, primarily epilithic, some epiphytic and metaphyton; single macrophyte species *Ruppia maritima*; ^{14}C methods for all components (Wetzel, 1964)
Phytoplankton			249.3	101.0	926	(56.8)	
Littoral algae			731.5	75.5	692	(42.5)	
Macrophytes			76.5	1.36	12	(0.7)	
					1630		
Marion, British Columbia	13.3	2.2					Softwater, oligotrophic lake; benthic algae, primary epipelic; O_2 techniques, from which net production was estimated (Efford, 1967; Hargrave, 1969; Gruendling, 1971)
Phytoplankton			21.9	0.29	8	(1.6)	
Littoral algae			109.6	11.3	310	(62.2)	
Macrophytes			49.3	6.5	180	(36.1)	
					498		
Lake 239, Ontario	56.1	10.5					Softwater, oligotrophic lake; probably underestimates since winter production is not included; benthic algae, primarily epilithic, macrophytes probably insignificant; CO_2 utilization methods (Schindler, et al., 1973)
Phytoplankton					823	(99.0)	
Littoral algae					8.1	(1.0)	
Macrophytes					N.D.		
					ca 831		
Lake 240, Ontario	44.1	6.1					(Same as above for Lake 239)
Phytoplankton					501	(98.2)	
Littoral algae					9.0	(1.8)	
Macrophytes					N.D.		
					ca 510		
Lawrence, Mich.	5.0	5.9					Hardwater, obligotrophic marl lake; benthic algae, primarily epiphytic on sparse submersed macrophytes; ^{14}C methods (Wetzel, et al., 1972)
Phytoplankton			118.9	2153	434	(25.4)	
Littoral algae			2003.	1977	399	(23.3)	
Macrophytes			240.8	4360	879	(51.3)	
					1712		
Wingra, Wisc.	139.6	ca 2					Large, shallow hardwater eutrophic lake; large littoral zone with dominant submersed macrophyte *Myriophyllum* and metaphytic mats of macroalga *Oedogonium*; ^{14}C methods for all components; mostly only summer values (McCracken, et al., 1974; Adams and McCracken, 1974; J. F. Koonce, personal communication)
Phytoplankton			1200.	1675.	4380.	(78.6)	
Methaphyton	(Summer, 1971)		3.0	4.2	11.1	(0.4)	
(*Oedogonium*)	(Summer, 1972)		5.5	7.6	19.9		
Macrophytes			320.5	447	1170	(21.0)	
					5581.		

Macrophytes

(0.4)

add under
(78.6)
(0.4)

Interactions Between Macrophytes and Phytoplankton. In recent years, there has been considerable interest in the relationship between algal and aquatic macrophyte growth. This interest has developed because of concerns about what the impact of reducing algal or aquatic macrophyte growth might be on subsequent algal or aquatic macrophyte growth. A number of investigations have suggested that an inverse relationship exists between development of phytoplankton and aquatic macrophyte populations (Kofoid 1903; Schreiter 1928; Hasler and Jones 1949; Hogetsu et al. 1960; Stangenburg 1968; Goulder 1969; Dokulil 1973) (Table 9). This relationship may exist because of shading by the aquatic plants (Hasler and Jones 1949; Dendy 1963; Westlake 1968; Goulder 1969), nutrient competition (Embody 1928; Weibe 1934; Bennett 1942; Fitzgerald 1969) or because the plants secrete inhibitory substances (Fitzgerald 1969). Other studies (Nichols 1973; Gasith et al. 1976; Lie 1978) suggest that aquatic macrophytes may enhance development of phytoplankton populations by mobilizing sedimentary phosphorus and releasing it to the water where phytoplankton assimilate the phosphorus.

Management of phytoplankton and aquatic macrophyte populations requires that water resource managers be able to predict how changes in phytoplankton and aquatic macrophyte densities interact. There are no quantitative data available that can be used to determine how phytoplankton density affects macrophyte density and vice versa. There is a strong need for quantitative data on phytoplankton and aquatic macrophyte biomass from a large number of aquatic systems representing a range of physical, chemical, and biological conditions in order to determine the relationship between phytoplankton and aquatic macrophyte biomass. This information would be especially useful to water resource managers because it would permit prediction of how these plants would respond to control. For example, if there is an inverse relationship between phytoplankton and aquatic macrophytes it might be possible to manage one group of plants to a level where phytoplankton and aquatic

Table 9.
Effect of dense aquatic plant growth on abundance of phytoplankton.
From Hasler and Jones, 1949.

		Plant-free 10^5 organisms per liter				Plant-filled 10^5 organisms per liter			
Year		1945		1946		1945		1946	
Pond ..		A	D	B	C	B	C	A	D
Month	Week								
July	1	99.00	15.40	9.13	2.07	71.00	4.80	4.39	10.97
	2	15.60	10.60	47.82	8.59	5.30	5.60	4.39	11.91
	3	21.00	2.70	28.06	15.86	2.90	5.40	2.70	2.56
	4	31.60	44.60	50.00	24.40	2.90	3.80	3.05	31.72
Aug.	1	8.40	26.00	90.28	13.42	3.20	2.00	1.95	5.61
	2	7.80	56.00	272.06	42.70	6.30	4.40	5.24	7.17
	3	130.00	56.00	394.06	9.42	3.80	1.90	8.27	1.70
	4	134.00	39.01	No sample		4.10	4.30	No sample	
Sept.	1	143.00	102.00	No sample		16.00	9.30	No sample	
	M	65.60	39.13	127.35	16.63	12.83	4.61	4.27	10.23

macrophyte problems are both minimized. However, if aquatic macrophytes support phytoplankton growth by releasing sedimentary phosphorus, it might not be possible to control algal biomass through reductions in nutrient inputs until aquatic macrophyte populations are controlled.

Effects on Invertebrates. There has been considerable interest concerning the effect of aquatic plants on invertebrate populations that are often the main food source for fish. Early workers (Forbes 1887; Reighard 1915; Baker 1918; Shelford 1918; Wilson 1924; Needham 1929, 1938; Surber 1930 Pate 1932, 1934; and many others) noted that there was often a considerable increase in the population of invertebrates when aquatic macrophytes were present. Lundbeck (1927) showed that in central European ponds, plant-free bottom areas supported 6.28 g of organisms/m^2; the submersed aquatic macrophytes 6.41 g/m^2; the emergent plants 8.29 g/m^2. Surber (1930) showed that snails were six times more abundant in weedy areas than non-weedy areas. In streams, the studies of Needham (1929) and Pate (1932, 1934) have shown that pools with weedy vegetation had between 17 and 38 times more invertebrates than bare pools.

The greater numbers of invertebrates in macrophyte areas has been attributed to the ability of plants to provide a substrate and shelter (Shelford 1918). Shelford (1918) noted that Elodea was an excellent plant for production of invertebrates. Andrews and Hasler (1943) noted that aquatic macrophytes with the most dissected surface area harbor the largest populations of invertebrates (Table 10). Martin and Shireman (1976) found that many invertebrates (450 g/kg vegetation) live in hydrilla. Welch (1952) noted that aquatic macrophytes provided a place of attachment to bryozoans, mollusks, annelids, and insects. Aquatic macrophytes may also increase invertebrate production indirectly by creating a favorable habitat. For example, by reducing wave action, macrophyte beds create important nursery areas for mosquitoes (Bishop and Hollis 1947; Beadle and Harmston 1958; Myklebust and Harmston 1962).

Table 10.

Number of organisms per kilogram of different types of aquatic plants. From Andrews and Hasler, 1943.

Most productive	*Ceratophyllum demersum* (coontail)	52,000 animals per kg. of plant
	Myriophyllum exalbescens (water milfoil)	29,000 animals per kg. of plant
Moderately productive	*Potamogeton pectinatus*	21,000 animals per kg. of plant
	Chara sp.	17-20,000 animals per kg. of plant
Less productive......................	*Potamogeton americanus*	18,000 animals per kg. of plant
	" *Richardsonii*	10,000 animals per kg. of plant
	" *amplifolius*	5,000 animals per kg. of plant
Poorly productive	*Vallisneria americana*	3,000 animals per kg. of plant

add ⊙ after kg.

Aquatic plants are an important source of food for invertebrates. Rawson (1955) showed that bottom fauna biomass (Figure 12) increased as lakes become shallower and more productive. Noonan (1979) showed that total number of zooplankton was directly related to algal biomass (Figure 26). These studies strongly suggest that systems of high primary productivity support more invertebrates than systems of low primary productivity.

Aquatic plants can enhance development of invertebrate populations, but can adversely affect the animals directly and indirectly. Toxins released by algae and macrophytes can kill invertebrates. Hutchinson (1975) reviewed the literature demostrating that both aquatic macrophytes and macroalgae of the family Characeae secrete inhibitory or insecticidal organic compounds that affect mosquitoes. Pennak (1973) also reviewed literature that showed organic secretions to inhibit or repel zooplankton. McLachlan (1970) reported that obstruction of light by plants caused a reduced diversity and biomass of benthic fauna. A major indirect effect occurs when plant decomposition reduces oxygen concentration below the tolerance levels of invertebrates.

There is a great deal of information on the relationships of aquatic plants to invertebrate populations, but most of the data are based on animal numbers or are only qualitative. There is, as of yet, no means by which water resource managers can predict the effect of aquatic plant growth on invertebrate populations. Biomass data on phytoplankton and aquatic macrophytes should be related to invertebrate biomass data from a range of aquatic systems. This should permit development of general equations to predict the impact of phytoplankton or aquatic macrophytes on invertebrate populations.

Fish. There has been considerable interest on relationships of plant and fish populations. Rawson (1955) showed that commerical fish production was inversely related to mean lake depth. His data strongly suggested that more productive lakes support more fish. However, as lakes become more productive, composition of fish communities may change, resulting in development of undesirable fish populations. Dense populations of aquatic plants can also reduce fish populations. Studies by Prescott (1932), Carl (1937), Prescott (1948), Mackenthun et al. (1948), Lefevre et al. (1952), and Shelubsky (1951) suggest that toxins released by blue-green algae have killed fish. When aquatic plants die and decompose, oxygen concentration is reduced, resulting in summer or winter fish kills (Tomlinson 1935; Sears 1936; Hutchinson 1936; Moore 1942).

Effects of aquatic macrophytes on fish populations are still debated. Early workers (Reighard 1915; Welch 1916, 1924; Baker 1918; Klugh 1926; Frohne 1938) noted that aquatic macrophytes supported many organisms eaten by fish. Klugh (1926), who reviewed much of the literature on relationships of invertebrates to aquatic macrophytes, concluded that the plants could be used as an index of fish production. Bailey (1978) reported that condition factors for bluegill and redear sunfish in Arkansas generally improved with removal of aquatic vegetation by grass carp. Colle and Shireman (1980)

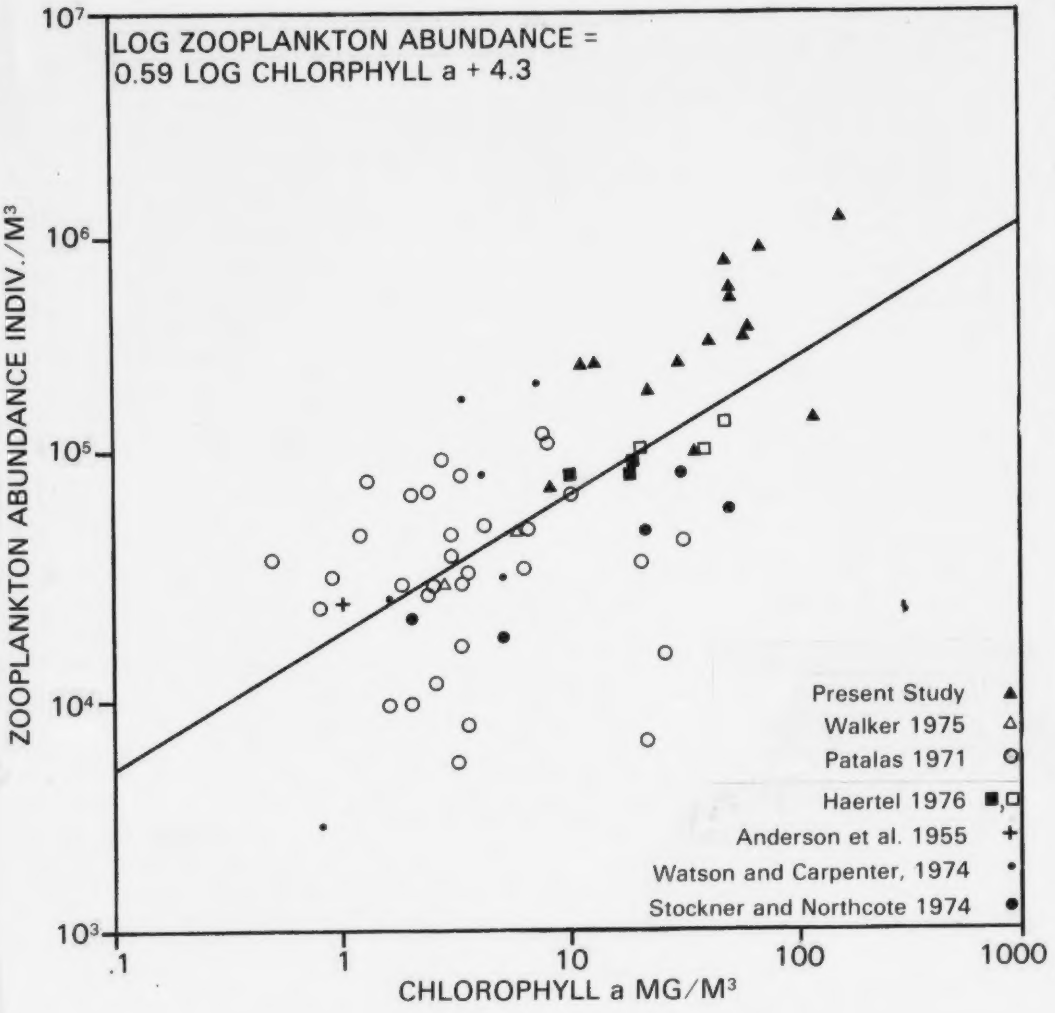

Figure 26. Lake mean zooplankton abundance versus mean chlorophyll a for data from the literature and this study. Values from Patalas (1971) include mean depth data from Brunskill and Schindler (1971) and chlorophyll a data from Armstrong and Schindler (1971). From Noonan, 1979.

found that condition factors of bass, bluegills, and redears were influenced by the amount of hydrilla in two Florida lakes. Harvestable largemouth bass had low condition values once hydrilla coverage was greater than 30%; however, smaller bass were not as adversely affected until cover exceeded 50%. Bluegill and redear condition and weight-length relationships were not affected by hydrilla until it occupied the majority of the water column. Cope et al. (1969) and Cope et al. (1970) found that bluegill and redear sunfish in Oklahoma ponds grew faster when submersed vegetation was controlled, and reasoned that a greater amount of food was available for growth in such situations. Ricker (1942) reported no association between bluegill growth rates and abundance of aquatic vegetation in a series of Illinois ponds. Buck and Thoit (1970) attributed delayed bass spawning to dense Potamogeton mats, which caused elevated pH (10.2). The fish spawned after vegetation was removed and pH values reduced. Stocking of Lake Lichen with grass carp caused alterations in fish species composition and number. Spawning grounds for some fish were destroyed as grass carp ate vegetation (Opuszynski 1972). Other studies (Hotchkiss 1941; Tilghman 1962; Wahlquist 1969; Gaevskaya 1969; and others) have shown that aquatic macrophytes provide sites for attachment of eggs and shelter for adult and larval animals, leading to increases in population size of some fish species.

Other studies suggest that aquatic macrophytes are detrimental to fish populations. Smith and Swingle (1941a, 1941b) and Rasmussen and Michaelson (1974) suggested that dense growths of aquatic macrophytes cause overcrowding by providing shelter to small fish. They also suggested that aquatic macrophytes sequester nutrients which could be used by phytoplankton, zooplankton, and fish. Bennett (1948) showed an apparent inverse relationship between fish yields and densities of aquatic macrophytes and suggested that macrophytes reduced fish production. Surber (1961) suggested dense stands of macrophytes interfered with fish production. Boyd (1967), however, noted that while many workers consider aquatic macrophytes detrimental to fish production, there are very little data that show any correlation between production and macrophytes biomass.

CONTROL METHODS IN THE SUNBELT

Floating Aquatic Weeds

Historically, floating aquatic weeds have caused problems in the Sunbelt states. They are restricted to this area primarily because they cannot survive prolonged freezing and ice cover. The most widely spread floating aquatic plant requiring control is the water hyacinth, which is believed to have originated in South America. It was first introduced into Louisiana and soon spread throughout the South.

Prior to 1950, chemical control of water hyacinth was accomplished by spraying the foliage with sodium arsenite, copper sulfate, and other desiccating mineral salts. Since 1950, water hyacinths have been controlled principally by 2,4-D (2,4-dichlorophenoxy acetic acid). Several salts, esters, and formulations of 2,4-D have been tested, and over 90% of the water hyacinths that are controlled chemically are with 2 to 4 lb acid equivalent/acre of the dimethlamine salt. The liquid 2,4-D formulation (Weedar 64) is registered for use on water hyacinths and has been granted fish and potable water tolerances. In 1977, it was estimated that approximately 300,000 lb of 2,4-D were used in Florida by state, federal, and local agencies responsible for water hyacinth control. Control programs in Florida cost an estimated $3.5 million and resulted in spraying of about 100,000 acres of water hayacinths (Haller 1976).

Because this herbicide formulation is a water-soluble liquid, application of 2,4-D to water hyacinth is simple and accomplished by several means, including hand-held spray guns, truck-mounted sprayers, and fixed wing or helicopter aircraft. Most water hyacinth spraying, however, is accomplished from airboats. Spray crews composed of an airboat and spray operator, can spray 10 to 20 acres of water hyacinth daily. The majority of water hyacinth spray programs require a 16-foot airboat, a 10-gallon per minute piston pump, and a saddle tank for holding the 2,4-D. The chemical is metered into the suction side of the pump by inserting orifice plates of various sizes into the the delivery line. The water portion of the spray mix is obtained through water ports at the back of the airboat. Typically, about 70 to 100 gallons of spray mix are applied with a handgun to each acre of water hyacinths. Spray mix (100 gallons for example) contains 99 gallons of water and one gallon (4 lb/gal) of 2,4-D. Most water hyacinth control programs keep the plant under maintenance control. Hyacinths are sprayed in isolated mats and in fringes of swamps and backwaters to maintain a tolerable number of plants because eradication is impossible in most watersheds, and a few plants help support a diversity of fish and wildlife.

Acreage of water hyacinth sprayed in a particular watershed varies from year to year. A 2- or 3-year succession of mild winters in the South permits vigorous water hyacinth growth the following summer and frequent and extensive control operations are often needed. Periods of heavy rainfall can raise water levels and flush water hyacinths from backwaters into flowing streams, lakes, and reservoirs where they must be controlled or they would fill the waterway.

Other floating aquatic plants which periodically require chemical treatment are water lettuce, Pistia sp., and duckweed. Water lettuce is found only in tropical areas, whereas duckweed is a cosmopolitan species occurring throughout the United States. In states that have functioning aquatic weed control programs, both water lettuce and duckweed are treated similarly to water hyacinths, but a different chemical (diquat) is used. Generally, 2,4-D does not kill water lettuce and duckweed. Diquat is substituted for 2,4-D in the spray mix and sprayed directly on the foliage. Other chemicals are used in some states for duckweed control, legally and illegally, depending upon the state registrations. Substituted ureas, triazines, and other herbicides applied at low rates (2 to 3 lb/acre) control duckweed very effectively particularly in small ponds.

Recent studies of dense water hyacinth infestations conducted at the University of Florida have delineated the environmental impact of this species on the aquatic ecosystem. Three 2-acre ponds were selected as test sites for detailed studies. Two ponds were chemically treated for hydrilla control, leaving the third pond as a check.

Water beneath hyacinth mats is devoid of life. Neither oxygen nor light penetrates beyond one-third meter, and anaerobic conditions prohibit fish survival. Natural organic turnover of the hyacinth mat (standing crop of 12 tons of dry matter per acre) leads to organic siltation rates of approximately 0.2 kg/m^2/yr. Chemical control and eradication of water hyacinths produced sediment accumulation of 0.6 kg/m^2/yr of treatment. On a longterm basis (10 years, for example), the natural untreated water hyacinth mat contributes 2 kg/m^2 dry matter to the sediment. Eradication of the mat produces 0.6 kg/m^2 of water hyacinth sediment plus whatever may be contribute by the phytoplankton in the remaining years after hyacinth control.

In conclusion, water hyacinths are the major floating weeds requiring treatment in the United States. Most water hyacinths are under maintenance control, and minimal spraying is required to keep them in check. The broadleaf herbicide 2,4-D is the most widely used to control water hyacinths and may be applied by hand-held spray equipment or from the air in major plant population areas. If the chemical control of water hyacinths was banned, there are no substitute control means, and vast water bodies in the South would become covered with water hyacinths.

Emergent and Ditchbank Weeds

Although often overlooked as a group, ditchbank and emergent weeds comprise a serious problem in irrigation and drainage districts, particularly in southern Florida and in the western United States. The problem with this group of weeds is their wide diversity. Saltcedar in the Southwest, grasses in general in the West, and *Melaleuca*, Brazilian pepper, willows, and various grasses in the Southeast create problems in irrigation ditches.

Economic data for the control of ditchbank weeds are difficult to assemble and are at best rough estimates. In 1977, Florida expended $8.2 million for ditchbank aquatic weed control. The U.S. Department of Agriculture (USDA) reported in 1968 that 27 states chemically treated 1.7 million acres of ditchbanks at an average cost of $20 per acre (total cost $34 million).

Few herbicides are labelled for ditchbank use. The generalization can be made, however, that dalapon is used for control of ditchbank and emergent grasses, whereas various formulations of 2,4-D are used to control broadleaf or dicotyledonous species. These chemicals are not effective against all ditchbank and emergent plants and those that are resistant to them cannot be controlled by currently approved chemicals. Because these species cause local problems, chemical industries consider them a minor problem and are not developing new, effective chemicals for their control. Consequently, weed control agencies increase application rates of dalapon and 2,4-D to high levels in order to control these plants and re-treat the areas more frequently.

Submersed Weeds

Submersed aquatic weeds are very difficult to control. The basic reasons are: 1) phytotoxic chemicals are diluted and degraded by water and aquatic organisms; consequently, it is difficult to get a toxic dose of the herbicide onto or into submersed aquatic weeds; 2) most waters in the United States have multipurpose uses and flow from one area to another. The same water, treated to control aquatic weeds, may be used by sportfisherman, for agricultural irrigation, sports, and ultimately as potable water; 3) a plethora of exotic aquarium plants have been introduced into the United States. Little is known about their life cycles, physiology, or biology. Without this knowledge, it is difficult to formulate treatment schedules.

Because of the diversity of submerged aquatic weeds, there currently are many chemical treatment methods varying from dragging a burlap bag of copper sulfate behind a rowboat to sophisticated invert emulsion applications from an airboat.

Many submersed aquatic weeds are treated by dissolving copper sulfate in ponds and lakes. Total volume treatment methods evolved from this traditional method to use of chemicals such as an endothall, diquat, diuron, acrolein, and various triazine herbicides. Two major problems became evident when total volume treatments were used: fish were frequently killed and/or the cost of chemical treatments became exorbitant. The use of copper sulfate was decreased due to its toxicity to fish, corrosiveness, and ineffectiveness in hard water where copper ions precipitate. Total volume treatment with diquat (for example, in 1 acre 10 feet deep) costs well over $300 for chemicals alone. Less expensive chemicals (diuron, triazines, etc.) often control submersed vegetation on a total volume basis more cheaply; however, these chemicals are persistent and water cannot be used for irrigation, etc., after treatment. Endothall and acrolein are toxic to fish at total volume concentrations that kill submersed weeds.

Although expensive short-term residual (1-3 weeks) herbicides are cleared for use in aquatic situations, endothall, diquat, organic-chelated liquid cooper, and 2,4-D are predominately used. Total volume treatments are used in irrigation ditches of the western states where acrolein or other contact phytotoxic chemicals are injected into the water from bridges or banks. A recent review of herbicides in flowing irrigation water has been published (Bowmer et al. 1979). Anhydrous ammonia has also been used successfully in western irrigation ditches for *Najas* and algae control. Basically, total volume treatment methods have been replaced by newer application methods, particularly in the southern states where aquatic weed problems have occurred for many years.

Deep-water injection techniques have evolved in an attempt to place phytotoxic chemicals in close proximity to or onto submersed weeds for more effective uptake, control, and cost reduction. Deep-water injection is used in some areas and has been used with success with invert or polymer carrier systems. Deep-water injection is conducted regardless of water depth from an airboat or other spray platform equipped with a pump capable of pumping tank mix or chemical directly into weighted hoses 12- to 15-feet long trailing behind the spray boat. Chemicals generally used in this application are endothall or diquat-copper complex combinations at roughly 4 to 8 gallons of chemical per acre. The ends of the trailing hoses are fitted with lead pipes to hold them down in the submersed weeds.

An additional modification, the invert system, allows placement of aquatic herbicides onto the target plants in slowly flowing water. Originally researched and developed for drift control, the invert system involves mixing an oil phase and an aqueous phase through a blending process into a mayonnaise-like homogenate. The homogenate is injected below the water surface and the droplets sink onto the submersed aquatic weeds. The droplets dissolve slowly, releasing the herbicide close to the target plant. The invert system has wide acceptance in the southern states, particularly for hydrilla control in Florida.

The basic mix generally used is (per acre):

Oil Phase	Water Phase
4 to 8 gal diesel fuel	30 to 40 gal water
2 to 3 gal inverting oil	2 gal diquat
	4 gal (weighting agent) liqud copper

This technique requires two tanks, an invert pump, and a below-surface injection system (weighted trailing hoses).

The previous disscussions of chemical treatment covered application of liquid formulations. Currently, two grandular formulations are widely used for hydrilla control [Hydout^R pellets - mono (N,N-dimethylalkylamine salt of endothall)] and for control of Eurasian watermilfoil [(Aquaclean^R granules - 2,4-D butoy ethanol ester (2,4-D BEE)].

The use of Hydout pellets for hydrilla control has increased in Florida and other states. Endothall pellets reduce the toxicity of the compound to fish and provide for good hydrilla control in areas where liquid herbicides have been less effective. The pellets are applied at rates of 100 to 300 lb per acre (22.4% active ingredient) depending upon water depth and degree of weed infestation. Application is generally accomplished from a granule fertilizer spreader on the bow of a boat. The pellets sink onto the weed mass and slowly dissolve, releasing herbicide. Aerial application of Hydout is used for very large scale treatments and is very effective.

Milfoil is effectively controlled with 2,4-D formulations, the most effective being 2,4-D BEE. TVA reservoirs have been experiencing milfoil problems for many years, and an increase in incidence of disease carrying mosquitoes was directly correlated to flood conditions and severity or extent of submersed weed infestations. Flood-water mosquitoes could be effectively controlled by draining standing water and regulating water levels, but biting insects (mosquitoes and tabanid flies) associated with aquatic weeds were controlled only by massive application of insecticides.

A large scale milfoil control program was initiated in several of the problem reservoirs in 1973. During the summer of 1974, over 400,000 pounds of 2,4-D granules were applied to milfoil. The operation was successful and currently milfoil is spot-treated and maintained by fluctuating water levels (Bates, personal communication, and Goldsby et al. 1978).

Herbicide pellets or granules are used extensively for control of certain aquatic plants. The advantages of using solid granules are ease and safety of application, simple application equipment, and release of the herbicide into the zone of the target plants. A particular disadvantage of using granules is the greater expense of manufacture. For example, the liquid endothall in the Hydout pellets costs approximately $10 per lb of active ingredient. After formulation, packaging and shipping, the pelletized formulation costs approximately $12 per lb active ingredient.

Currently, the most extensive submersed aquatic weed control programs are in Florida, Texas, and Louisiana where hydrilla is a problem. Hydrilla infests virtually every watershed in Florida, and only a small portion of the total acreage of hydrilla is treated annually due to economic restrictions. For example, Orange Lake in 1977 contained 10,000 acres of hydrilla, but only 200 acres were controlled by mechanical or chemical means. It is estimated that expenditures for hydrilla treatments in Florida in 1977 were $9.1 million. These treatments covered approximately 45,000 acres at an average cost of $200 per acre. Approximately 400,000 lbs. of phytotoxic chemicals are placed in Florida's waters each year for submersed weed control.

Chemical weed control has been emphasized in this report because governmental agencies spend more than $20 million each year on aquatic weed control in Florida, at least 90% of the expenditure is for chemical aquatic weed control.

WATER LEVEL FLUCTUATION

Habitat alterations, such as water drawdowns, pond liming, fertilization disking, etc., have been used with various degrees of success for vegetation control. Water level fluctuations on large reservoirs have generated great interest in the past decade. This method can be used in many waterways, especially southern reservoirs, but fluctuation schedules must be established individually for each target species. The following are examples where water fluctuation has been used to manage aquatic vegetation:

1. Ross Barnett Reservoir in Jackson, Mississippi, was drawndown several feet during the winter of 1978 to control *Najas* species;

2. Tennessee Valley reservoirs are being fluctuated regularly for Eruasian watermilfoil control,

3. Louisiana has drawndown lakes during winter control of Egeria, Cabomba, and Hydrilla species;

4. Florida has drawndown Lake Myacca and Lake Ocklawaha for Hydrilla control;

5. A 5-foot increase in water level was largely responsible for 90% control (for 2 years) of hydrilla in Orange Lake, Florida, in 1977.

Water-level fluctuation has controlled some aquatic weeds very effectively. It appears that properly timed water manipulation is a key factor, but research has been sporadic and often improperly monitored.

Hydrilla control by drawdown is similar to chemical control because only the above-ground portions of hydrilla are killed, leaving viable tubers in the hydrosoil to cause reinfestation. Drawdowns timed with respect to formation and germination of hydrilla tubers offer partial management of tuber populations.

Figure 27 outlines a drawdown schedule that provides hydrilla control in northern Florida. Drawdown 1 is considered optional because its primary purpose is to stimulate tuber germination after water levels are returned to normal the following summer. Hydrilla tubers germinate only once, and disappear after producing a new plant. The optional drawdown can be conducted any time in late winter or early spring, and its duration depends on local conditions. The drawdown must be long enough to dry the top few inches of the hydrosoil.

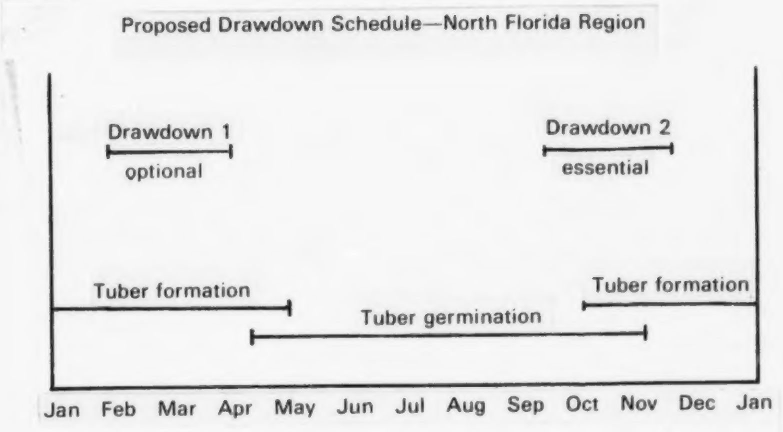

Figure 27. A drawdown scheme which will provide hydrilla control in North Florida.

Drawdown 2 is essential because it kills hydrilla plants which have sprouted from tubers and prevents tuber formation the following winter. This drawdown should be started before October (before new tubers are formed) and should be continued until water temperatures drop to around 55 F. This inhibits tuber germination and the establishment of new plants, thus preventing new tuber formation. Limited research indicates that after one year of both drawdowns, followed by two years of the essential drawdown, it will be possible to proceed with the essential drawdown every other year.

Water-level fluctuation for aquatic weed control would have greater potential use if more were known about the life cycles of target species and their responses to water-level changes. Not only could detrimental plants be controlled, but growth of beneficial plants could be encouraged. For example, Panicum sp. is a beneficial plant for fish production. This plant spreads and grows rapidly in deep water when water levels are reduced.

MECHANICAL CONTROL

Mechanical control is not stressed in this report because it is not widely used and in general is very expensive, producing only short-term effects. Extensive research efforts have been made to develop economical harvesting and other mechanical control methods, but most aquatic vegetation is 92 to 96% water, which makes it expensive to harvest and place on shore. Nevertheless, mechanical equipment of various types are available and include the following:

1. Cutters which cut or free submersed and floating vegetation, permitting it to flow downstream;
2. Harvesting systems which cut the vegetation, lift it from the water, and place it on shore;
3. Dredges and pumps designed to remove aquatic plants, organic detritus, and plant roots from the hydrosoil;
4. Draglines and backhoes.

Advantages of each of the systems are apparent. Most people, however, prefer to harvest aquatic weeds in order to remove nutrients in the plants. Studies in Florida have shown that mechanical harvesting of submersed vegetation removes an insignificant amount of nutrients and that damage to fisheries could result. Small sport and forage fish species live in submersed vegetation and become entrapped in vegetation as it is harvested. Recent studies have shown that 32% of a fish population can be removed by a single mechanical harvesting operation (Haller et al. 1980). Generally, very little mechanical harvesting or control of aquatic weeds is conducted in the South due to high cost.

BIOLOGICAL CONTROL

It is widely held that exotic weeds become problems in their new habitats because they lack natural predators or controls which co-evolved with them in their native habitat. Submersed weeds are spreading most rapidly in the south and are by far the most expensive to control. Prospects for devvelopment of new, inexpensive herbicides or more efficient mechanical harvesting equipment within the immediate future are not great. Consequently, more effective weed control programs in the field of biological control are needed.

Discovery and successful widespread use of the alligator-weed flea beetle (Agasicles hygrophila Selman and Vogt) for control of alligator-weed in the 1960s have provided impetus to study biological control of other aquatic plants (Maddox et al. 1971; Spencer and Coulson 1976). Three insects have been studied and released on water hyacinths in the United States, and a host-specific fungal pathogen (Cercospora rodmanii Conway) is currently being developed (Freeman 1977; Charudattan 1979).

There are hundreds of thousands of insect species in the world, but relatively few live in the submersed aquatic habitat where hydrilla thrives. The search for insect biocontrols for hydrilla has not yet yielded significant insect candidates (Zeiger, personal communication). Recently, a fungal pathogen (Fusarium culmorum) was isolated from plants from Holland and is a promising organism for hydrilla control. Extensive field studies were planned to begin in 1979 (Charudattan 1979).

Biological control studies of hydrilla with snails (Marisa cornuaretis L.) have been extensively studied in Florida. It was found that Marisa was not a significant biological control because it was temperature sensitive, and very high stocking densities were required (Blackburn et al. 1971).

Several species of fish have been considered as candidates for the biological control of submersed aquatic weeds (Blackburn et al. 1971; Legner et al. 1975). The Chinese Grass Carp or White Amur (Ctenopharyngodon idella Val.) recieved most attention for aquatic weed control in the South.

The grass carp was introduced into the United States in the 1960s and was first evaluated for aquatic weed control in Arkansas and Alabama. Arkansas has been the leading state in the use of carp for weed control: its major waters have been stocked with excellent control of many aquatic weed species. In other states, fishery biologists and environmental groups have proceeded with caution, and the grass carp has been banned from many southern states.

The first field research in Florida was initiated in three small (0.08 ha) earthen ponds in central Florida in 1971. This nonreplicated study (complete in January 1973) showed that stocking rates of 50 grass carp/ha controlled hydrilla without catastrophic effect on the aquatic environment (Haller and Sutton 1976).

Further major research was undertaken by the Florida Department of Natural Resources and the Florida Game and Fresh Water Fish Commission in 1972. Four natural ponds in widely scattered geographical locations were stocked with grass carp for collection of baseline data after one year. The diverse interpretation of the results of these nonreplicated studies has become widely known among the world's fishery scientists (Beach et al. 1976; Gasaway and Orda 1978).

In 1974, six lakes (each more than 50 ha) and one reservoir (2000 ha) were stocked with grass carp to further determine their weed control capabilities and potential environmental impact. This research is incomplete as three of the lakes have become weed-free, and three other lakes still contain hydrilla infestations. In one lake, apparent lack of biocontrol resulted from low grass carp populations (Colle et al. 1978). Restocking programs have begun on the remaining vegetated lakes and the reservoir.

Due to unpredictable results obtained with grass carp, and unanswered questions concerning its possible impact on sport fish populations, there remains considerable controversy among biologists with regard to widespread use of the fish in hydrilla control programs.

Currently, the State of Florida allows private possession of grass carp by individuals with weed problems in lakes (10 ha or less) that are not connected to other water bodies. Stocking is permitted only in private waters that meet specific criteria (size, weed problems, and lack of infall or outfall) and the grass carp is currently in use in golf course ponds, fishery ponds, and waters of similar nature.

Widespread application of the grass carp to solve hydrilla problems in large lakes has been deferred until further studies are conducted. The problem remains: hydrilla continues to spread and current control measures are expensive.

IMPACT OF CONTROL METHODS

Nutrient Reduction Techniques

1. Reduction of Plant Nutrient Inputs. Studies on natural lakes (Edmondson 1961, Sakamoto 1966; Vollenweider 1968, 1969; Shannon and Brezonik 1972; Schindler et al. 1973; Dillon and Rigler 1974a, 1974b; Jones 1974; Kirchner and Dillon 1975; Jones and Bachmann 1976; and others) have shown that input of plant nutrients is an important determinant of lake nutrient concentrations and plant biomass, particularly phytoplankton biomass. For this reason, it has been assumed that reduction in nutrient inputs would reduce excessive growth of phytoplankton and aquatic macrophytes. Edmondson (1966, 1969, 1970, 1972a), Michalski and Conroy (1973), and Schindler (1975) have shown that lakes respond to nutrient reductions: their studies have shown significant reductions in nitrogen and phosphorus concentrations. The biomass of phytoplankton as measured by chlorophyll a concentrations was also reduced. In general, there was an overall improvement in water quality.

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Although some lakes have responded to nutrient reduction programs almost immediately, lakes with a long history of high-nutrient loading rates have often failed to respond or have recovered very slowly (Alhgren 1972; Bjork 1972; Larsen et al. 1975). Schindler (1976) suggested that this difference in response time is related to the degree to which bottom sediments are saturated with nutrients. He suggested that the sediments can release nutrients into the water for long periods of time, thus delaying recovery of lakes that have recieved long-term, high-nutrient loading. Studies by Lamarra (1975a, 1975b), Gasith et al. (1976), and Lie (1978) have shown that bottom-feeding fish and rooted aquatic macrophytes can also enhance recycling of nutrients from bottom sediments.

2. Reduction of Internal Nutrient Recycling. Bottom sediments of an aquatic environment represents a potentally significant plant nutrient source, and sediment removal by dredging is often advocated. Bengtsson et al. (1975) reported that removal of sediments from Lake Trummen, Sweden, minimized nutrient recycling. They noted that phosphorus, nitrogen, silica, and phytoplankton concentrations declined, while phytoplankton diversity and water clarity increased.

Dredging to reduce nutrients does not always result in water quality improvement. In fact, water quality can deteriorate significantly during dredging operations. Peterson (1979) reviewed the literature on the impact of dredging on the aquatic environment and noted that dredging caused release of toxic substances and nutrients from sediments, increases in water turbidity, depletions of oxygen, and changes in pH and temperature. Many of these effects, however, last only a short time.

Another method of reducing nutrient recycling and nutrient concentrations in the aquatic environment is the application of chemicals that precipitate nutrients. Aluminum, iron, calcium, zirconium, and lanthanum (Gahler 1969) have been used to remove nutrients from water. Funk and Gibbons (1979) recently reviewed some of the literature concerning effects of nutrient inactivation techniques on the aquatic environment. They noted that reductions in total phosphorus, ammonia, and Kjedldahl nitrogen and iron concentrations, as well as algal standing crops, have occurred following chemical treatment. Improvements in water clarity and hypolimnetic oxygen concentrations have also been reported. However, Funk and Gibbons (1979) noted that these effects were not always consistent among lakes. They cited a Cooke and Kennedy (1977) report that chemical treatments were only partially successful in two of the lakes observed in their study. Cooke and Kennedy (1977) suggested that macrophytes and bottom fauna mediated release of nutrients from sediments, thus reducing the effectiveness of nutrient precipitation. Funk and Gibbons (1979) also noted that there were very little data concerning effects of chemical treatments on fauna of the lakes.

Artifical aeration has often been used to prevent regeneration of plant nutrients from sediments during anaerobic conditions. It was thought that aeration could prevent development of algal growth by preventing regeneration of nutrients from the sediments. Fast (1979) recently reviewed the literature on aeration and found that nutrient concentrations, particularly that of phosphorus in deep water, nearly always declined during destratification of lakes with aerators. Studies by Robinson et al. (1969) and Haynes (1971) have shown that destratification of lakes results in reduction of algal concentration. However, there is also evidence that destratification can result in increased algal numbers, particularly green algae (Hooper et al. 1952; Robinson et al., 1969; Haynes 1971; Ridley 1971). Fast (1979) noted that circulation of lake water could raise the temperature of deep-water sediments. With increased temperature and oxygen, Fast suggested these sediments could be colonized by benthic organisms and that their activities (Lee 1970; Brinkhurst 1972; Davis 1974) could increase nutrient exchange rates. Fast (1979) noted that there is no clear evidence that aeration reduces nutrient concentration within the euphotic zone of lakes. In fact, he showed that there is evidence that destratification upwells nutrients to surface water.

Aeration of lakes does have important effects on other components of the aquatic environment. Aeration of anaerobic waters reduces the concentrations of iron, maganese, nitrogen, and sulfur (Irwin et al 1967; Wirth and Dunst 1967; Symons et al. 1970; Haynes 1971). In addition, aeration can increase the oxidation rate of organic matter in bottom sediments and the water column (Mercier 1955; Fast 1971). Fast (1971) found that aeration resulted in rapid invasion of deep-water sediments by benthic organisms and an increase in the vertical distribution of zooplankton. Aeration of waters can also increase depth distribution of fish (Gebhart and Summerfelt 1975; Brynildson and Serns 1977). Despite all the beneficial effects noted for aeration, Fast (1979) reported that not all have been thoroughly documented, and in some lakes, beneficial effects noted in other lakes do not always occur.

Sediment exposure and desiccation has been suggested as a method for reducing internal recycling of nutrients because oxidation of the sediment surface should retard release of nutrients (Mortimer 1941; 1942) and increase the binding capacity of sediments (Fitzgerald 1970). Sediment exposure can also curb sediment nutrient release by physically stablizing the upper zone of flocceluent sediments (Lee 1970). However, other studies (Sneisko 1941; Neess 1946; Davis and Lucas 1959) suggest sediment desiccation will accelerate microbial mineralization of organic matter, thus making inorganic nutrients available for plant growth upon reflooding. Drawdown and reflooding are used widely to increase plant growth in marshes and fish culture ponds.

Effects of Herbicides on the Aquatic Environment

For this report, we will concentrate on the five most widely used aquatic herbicides and their various formulations. Federal Laws, particularly the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), have greatly reduced the number of herbicides used in aquatic ecosystems. All chemicals currently used for aquatic weed control must have either federal registration (full or interim) and/or must be used on a large enough basis to warrant state special need registration.

Because of these restrictions, the predominant herbicides are those sold in quantities great enough to offset the expense (toxicology, residue, metabolism, and ecological studies) required for registration. Consequently, various formulations of copper, endothall, Dalapon, 2,4-D, and Diquat constitute the vast majority of aquatic herbicides used in the United States (Haller 1979).

It should also be stated that the products mentioned in this section have full registration, or at least have interim tolerances established for water and fishes. These compounds are, as far as is known, safe to use in the aquatic environment. Nevertheless, effects of these compounds on water quality, non-target organisms, and other selected parameters will be reviewed in this section.

2,4-D (2,4-dichlorophenoxy) Acetic Acid

2,4-D is the parent acid used to make many formulations (salts, amines, esters, etc.). These formulations have differential herbicidal activity on different species of weeds and also have different residue, toxicology, and metabolite characteristics.

The dimethyl amine salt of 2,4-D is a wide-spectrum broadleaf weed killer. It is not effective on grass species (monocots). It is most widely used in water hyacinth control (2 to 4 lb/acre) and for control of ditchbank dicots (6 to 10 lb/acre) (brush, etc.). The butoxy-ethanol ester of 2,4-D, (2,4-D BEE), often formulated on granules, is used for control of waterlily species (*Nuphar*, *Nymphaea*) and Eurasian watermilfoil. Application rates vary between 10 and 40 lb active ingredient per acre, depending upon species.

Residues in Water and Soil. The half-life of 2,4-D in water varies with environmental conditions and habitat sprayed. In general, high water temperatures, dissolved oxygen, dissolved organic matter, etc., cause rapid degradation for all herbicides.

2,4-D dimethyl amine sprayed on foliage of water hyacinths or on ditchbank weeds results in very little 2,4-D contamination of water. It has been shown that of 4 lb 2,4-D sprayed on an acre of water hyacinth, only 20 to 40% of the chemical is detected in the water, presumably the result of spray runoff from plant leaves. Thus, at the rate of 4 lb/acre, between 1 and 2 lb might be detected in the water, but the concentration would vary with depth. In a slow-flowing situation (e.g. canal), it is possible that 2,4-D in water would be undetectable due to the chemical being applied adjacent to water but not over it.

Residues of 2,4-D dimethyl amine applied at normal rates of 2 to 4 lb/acre dissipate from water to non-detectable levels in 10 to 20 days. Rate of dis- sipation or degradation of 2,4-D from hydrosoil is similar to water and varies depending upon environmental conditions (Tables 11 and 12). Breakdown of 2,4-D in aquatic systems is apparently due primarily to microbial metabolism (Frank and Comes 1967; Schultz 1973).

Table 11.
Residues of the dimethylamine salt of 2,4-D in water (mg/l), hydrosoil (mg/kg) and fish (mg/kg) from ponds in Florida and Georgia treated with 2.24, 4.48 and 8.96 kg 2,4-D per hectare. From Schultz and Gangstad, 1976.

Pond	Rate kg/ha	Depth m	Temp. C	Time Days	Water mg/l	Hydro-soil mg/kg	Fish mg/kg
Florida	2.24	1.3	34	01	0.025	0.005	0.080
	4.48	1.0	31	01	0.155	0.014	0.048
	8.96	1.2	31	01	0.312	0.033	0.005
Georgia	2.24	1.3	27	01	0.025	0.018	0.005
	4.48	0.9	29	01	0.233	0.024	0.014
	8.96	1.2	30	01	0.657	0.026	0.022
Florida	2.24	1.3	30	03	0.005	0.005	0.005
	4.48	1.0	30	03	0.172	0.014	0.005
	8.96	1.2	31	03	0.345	0.046	0.005
Georgia	2.24	1.3	29	03	0.087	0.008	0.005
	4.48	0.9	32	03	0.390	0.018	0.005
	8.96	1.0	30	03	0.692	0.040	0.005
Florida	2.24	1.3	31	07	0.005	0.005	0.005
	4.48	0.9	31	07	0.048	0.010	0.005
	8.96	1.2	31	07	0.025	0.008	0.005
Georgia	2.24	1.3	26	07	0.059	0.010	0.005
	4.48	0.9	27	07	0.400	0.018	0.005
	8.96	1.0	29	07	0.395	0.042	0.005
Florida	2.24	1.3	32	14	0.005	0.005	0.036
	4.48	1.0	32	14	0.005	0.010	0.005
	8.96	1.2	31	14	0.005	0.013	0.043
Georgia	2.24	1.3	28	14	0.027	0.005	0.005
	4.48	0.9	31	14	0.008	0.005	0.005
	8.96	1.0	30	14	0.050	0.005	0.005
Florida	2.24	1.3	30	28	0.005	0.005	0.005
	4.48	1.0	32	28	0.005	0.007	0.005
	8.96	1.2	30	28	0.005	0.005	0.005
Georgia	2.24	1.3	27	28	0.005	0.006	0.005
	4.48	0.9	31	28	0.005	0.005	0.005
	8.96	1.0	30	28	0.005	0.005	0.010

A literature review concerning herbicide residues or toxicology is difficult due to the tremendous variation in data from small-pond studies, laboratory studies, and large-scale studies done under non-standardized conditions. Residue data vary from 1 day to 6 months after treatment, and authors often do not list pertinent environmental conditions in their reports.

Large scale field tests are probably more useful in determining the effect of herbicides on the aquatic environment. Joyce and Sikka (1977) presented data on the use of 2,4-D dimethyl amine in a major hyacinth control program.

The St. Johns River is a slow moving Florida river approximately 300 miles long with a long-standing water hyacinth problem. Joyce and Sikka (1977) collected water samples at nine locations in the river and compared the amount of 2,4-D applied to its residue in each location (Figues 28 and 29). Several hundred pounds of 2,4-D were applied to shoreline marshes and backwaters of the river over 6 months. The maxium concentration of 2,4-D detected in the river was 1.4 parts per billion (ppb).

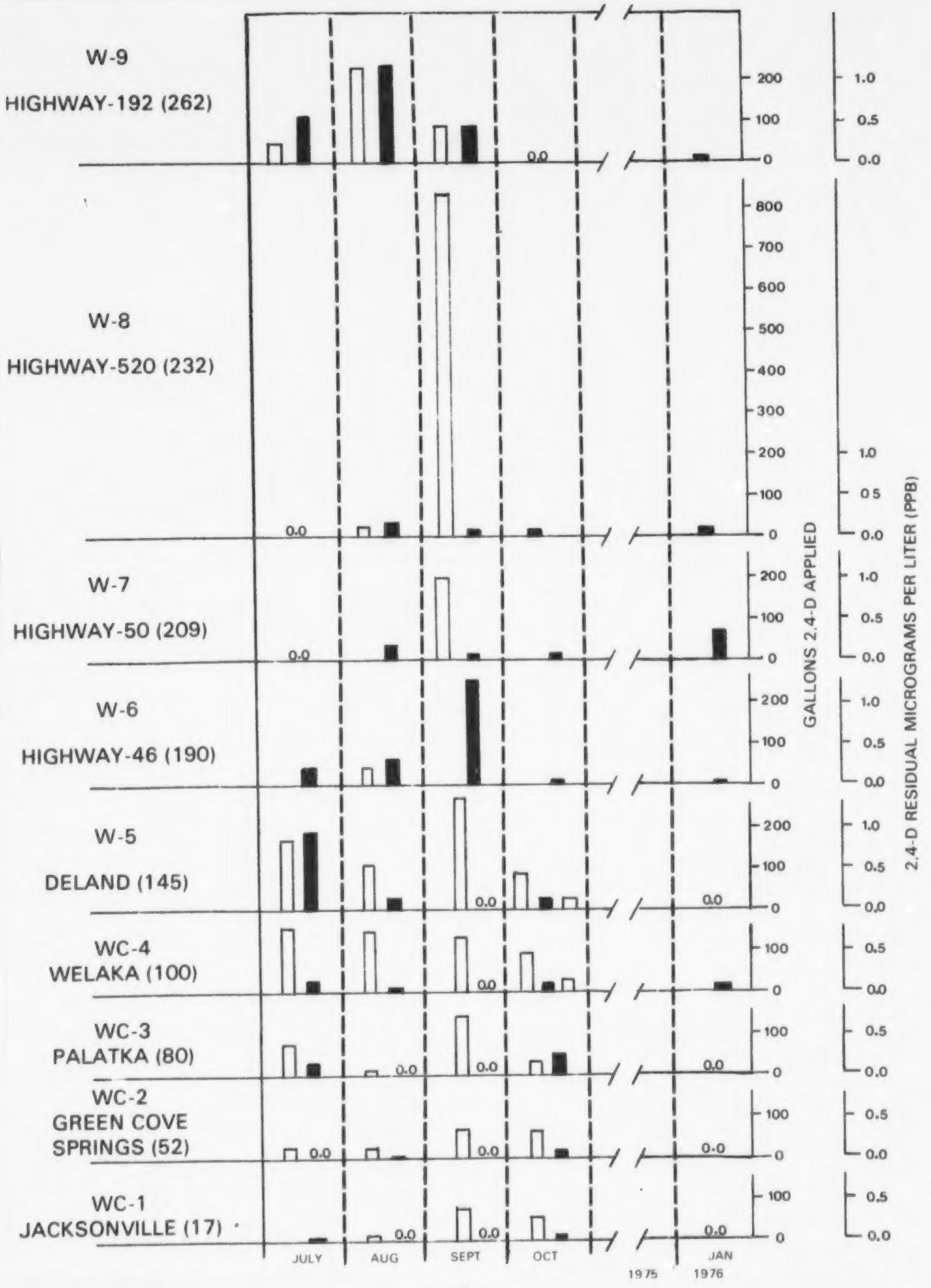

Figure 28. Comparison of 2,4-D residual in water and the number of gallons of 2,4-D applied monthly in the St. Johns River, Florida. Open bars represent cumulative gallons of 2,4-D, closed bars represent 2,4-D residual in ppb, and ( ) represents the distance upstream from mouth in river miles. From Joyce and Sikka, 1977.

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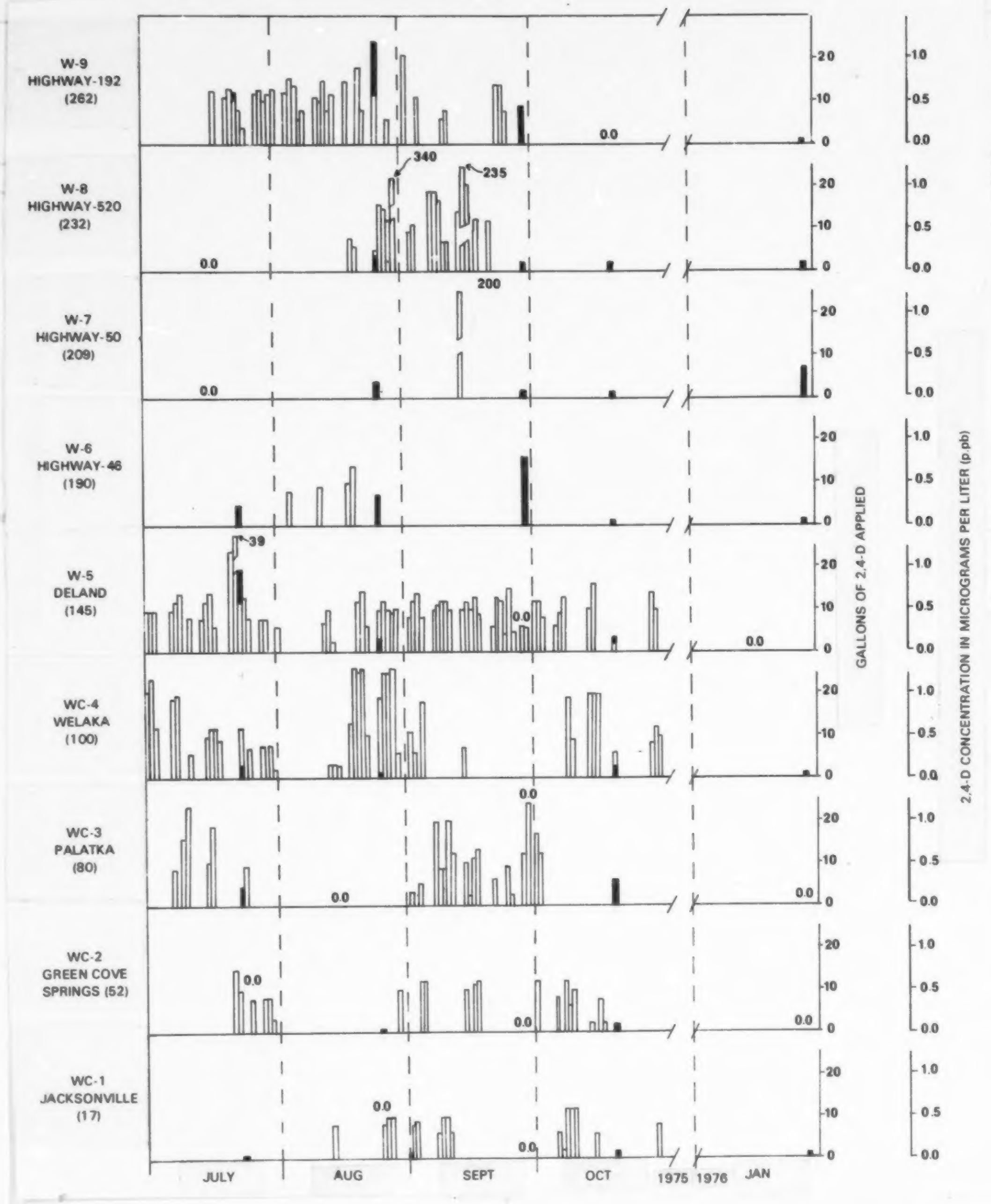

Figure 29. Comparison of 2,4-D residual in water and the number of gallons of 2,4-D applied daily in the St. Johns River, Florida. Open bars represent number of gallons of 2,4-D applied, closed bars represent 2,4-D residual in ppb, and ( ) represents distance from mouth in river miles. From Joyce and Sikka, 1977.

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Table 12.
Physical, chemical and toxicological properties of some aquatic herbicides. From Newbold, 1975.

Herbicide	Chemical formula	Mode of action	Persistence in muds	Breakdown in water	Form	Treatment level (mg/l)	Levels of toxicity to fish LC_{50}	Levels of toxicity to Daphnia LC_{50}
Dalapon	2,2-dichloroproprionic *acid*	contact/ *translocation*	10 to 60 *days*	2 to 3 *days*	Liquid	1.0	350 mg/l, 24 h *Salmo gairdnerii* (Richardson) (Alabaster, 1969)	6-0 mg/l, 48 h *Daphnia magna* (Straus) (F.W.P.C.A., 1968)
2,4-D amine salt	2,4-dichlorophenoxy-acetic acid (amine salt)	translocation	1 month	4 to 6 weeks	Liquid		250 mg/l, 24 h *Salmo gairdnerii* (Richardson) (Alabaster, 1969)	> 100 mg/l, 48 h *Daphnia magna* (Straus) (Crosby & Tucker, 1966)
Diuron	3-(3,4-dichlorophenyl)-1,1-dimethylurea	translocation inhibits photosynthesis	4 months	3 months	Liquid	0.25-0.5	4.03 mg/l, 48 h *Salmo gairdnerii* (Richardson) (F.W.P.C.A., 1968)	1-4 mg/l, 48 h *Daphnia palex* (De Geer) (F.W.P.C.A., 1968)
Asulam	methyl(4-aminobenzene-sulphonyl)carbamate	translocation, inhibits cell division	Not known	2 weeks	Liquid		5200 mg/l, 24 h *Rasbora heteromorpha* (Duncker) (Alabaster, 1969)	Not known
Copper sulphate	$CuSO_4$, $5H_2O$		Not known	Not known	Crystal	0.2-1.0	0.14 mg/l, 48 h *Salmo gairdnerii* (Richardson) (Shaw & Brown, 1974)	Not known
Maleic hydrazide	6-hydroxy-3-(2H)-pyridazinone	Translocation suppresses growth, inhibits cell division	Soils 1 to 8 weeks	Not known	Liquid	1.0	Information insufficient	Not known, likely to be very toxic (Greulach *et al.*, 1961)
Maleic hydrazide/ 2,4-D								
Maleic hydrazide/ 2,4-D/ chlorpropham	maleic hydrazide/2,4-D/N-(3-chlorophenyl) carbamate	mitotic poision						

A large-scale submersed weed control operation was conducted in 1966 in TVA lakes where dramatic increases in Eurasian watermilfoil hindered navigation and power production. Propogation of Anopheles mosquitoes necessitated use of larvicides to control mosquitoes. A report (Anonymous 1961) stated that "watermilfoil, like some other submersed aquatics, offers an extremely favorable late season habitat for Anopheles quadrimaculatus and other permanent pool mosquitoes."

Smith and Isom (1967) reported that 888 tons of 20% 2,4-D BEE granules were applied to 8,000 acres of milfoil in seven TVA reservoirs. The treatments spanned a 352-mile main channel distance and were applied between March and December 1966.

Fish mortalities, differences in pre- and post-treatment populations of Hexgenia, and toxic effects on benthic fauna did not occur. Smith and Isom (1967) concluded that high application rates (up to 100 lb/acre in flowing water) of 2,4-D BEE did not produce adverse effects on aquatic fauna or water quality (Tables 13, 14, and 15).

Table 13.
Analyses of water samples, Watts Bar Reservoir, Gordon Branch Embayment. From Smith and Isom, 1967.

						Alkalinity		2,4-D	
Station	Date	Time (FS)	Temp (°F)	Dissolved Oxygen (Mg/l)	pH	Phenol (Mg/l)	Total (Mg/l)	BEE (µg/l)	Acid (µg/l as BEE)
(Surface samples collected prior to 2,4-D application)									
A	3-17-66	0505	55.7	10.0	8.1	0.0	70.1	—	—
B	3-17-66	0825	56.3	10.2	8.6	1.4	35.9	—	—
C	3-17-66	0835	56.7	11.4	7.8	0.0	33.4	—	—
D	3-17-66	0840	56.2	10.9	8.1	0.0	34.9	—	—
E	3-17-66	0855	56.4	11.4	8.2	0.0	34.5	—	—
(Surface samples collected 1 hour after 2,4-D application)									
A	3-18-66	0920	55.4	11.7	8.8	6.0	94.0	<1	<1.45
B	3-19-66	0945	56.1	10.4	7.8	0.0	35.0	<1	<1.45
C	3-19-66	0955	37.2	11.4	8.3	0.0	34.0	<1	<1.45
D	3-19-66	1000	53.0	8.8	6.7	0.0	19.5	37	<1.45
E	3-19-66	1020	56.1	11.4	8.4	1.0	32.5	6	<1.45
(Surface samples collected 4 hours after 2,4-D application)									
A	3-18-66	1220	59.0	12.8	9.1	18.0	92.0	<1	<1.45
B	3-19-66	1230	61.6	10.8	7.9	0.0	35.0	6	<1.45
C	3-19-66	1240	59.3	11.5	8.5	2.0	36.0	<1	<1.45
D	3-19-66	1245	62.4	12.6	8.6	3.0	36.0	<1	<1.45
E	3-19-66	1305	56.8	10.4	8.5	1.5	35.0	2	<1.45
(Surface samples collected 8 hours after 2,4-D application)									
A	3-18-66	1550	64.4	11.6	8.7	5.0	91.0	<1	<1.45
B	3-19-66	1510	64.7	10.4	8.2	0.0	36.0	<1	<1.45
C	3-19-66	1515	62.2	11.4	8.5	2.0	38.0	<1	<1.45
D	3-19-66	1520	67.2	10.4	8.6	3.0	35.0	<1	<1.45
E	(Sample missing)								

Diquat [6,7-dihydrodipyrido (1,2-a:2',1'-c)pyrazinediium ion]

Diquat is probably the most widely used herbicide for submersed aquatic weed control in the United States. It is effective at a rate of 2 ppm or less on hydrilla, Elodea, milfoil, pondweeds, algae, and various other species. There is only one formulation of diquat, the liquid diquat dibromide.

The chemical is a contact weed killer, phytotoxic to plant tissue, and kills by interfering with photosynthesis. In aquatic systems it is used largely for control or submersed plants, but also is used in some instances on duckweed, water hyacinth, water lettuce, or ditchbank grasses.

This discussion will emphasize use of diquat on submersed plants.

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Table 14.

Analyses of water samples, Guntersville Reservoir, Vicinity of Comer Bridge. From Smith and Isom, 1967.

						Alkalinity		2,4-D	
Station	Date	Time (CS)	Temp. (°F)	Dissolved Oxygen (Mg/l)	pH	Phenol (Mg/l)	Total (Mg/l)	BEE (µg/l)	Acid (µg/l as BEE)
		(Surface samples collected prior to 2,4-D application)							
A	3-29-65	1120	57.0	9.2	7.7	0.0	37.0	3	<1.45
B		1110	57.0	—	—	—	—	<0.5	<1.45
C		1101	57.0	—	—	—	—	7	<1.45
D		—	60.0	8.8	7.5	0.0	39.4	7	<1.45
E		—	—	—	—	—	—	6	<1.45
F		1215	59.0	—	—	—	—	7	<1.45
		(Surface samples collected 1 hour after 2,4-D application)							
A	4-5-66	0835	56.5	8.8	7.2	0.0	36.0	91	<1.45
B		0845	56.0	8.7	7.6	0.0	41.0	157	<1.45
C		0855	56.3	8.6	7.4	0.0	41.0	36	<1.45
D		0925	56.0	9.0	7.6	0.0	36.0	5	<1.45
E		0910	56.0	8.6	7.6	0.0	38.0	5	<1.45
F		0905	56.0	8.6	7.7	0.0	39.0	3	<1.45
		(Surface samples collected 4 hours after 2,4-D application)							
A	4-5-66	—	—	—	—	—	—	—	—
B		—	—	—	—	—	—	—	—
C		—	—	—	—	—	—	—	—
D		1220	57.0	9.2	7.6	0.0	36.0	8	<1.45
E		1230	58.0	9.3	7.6	0.0	37.0	130	<1.45
F		1240	57.5	9.4	7.8	0.0	36.0	19	<1.45
		(Surface samples collected 8 hours after 2,4-D application)							
A	4-5-66	1455	56.5	8.6	7.4	0.0	39.0	2	<1.45
B		1505	57.0	8.5	7.5	0.0	39.0	<1	<1.45
C		1515	57.0	8.5	7.5	0.0	39.0	4	<1.45
D		1535	58.0	9.7	7.8	0.0	38.0	18	<1.45
E		1530	58.0	10.1	8.0	0.0	36.0	64	<1.45
F		1525	58.0	9.9	7.9	0.0	38.0	21	<1.45

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Table 15.

2,4-D analyses — Watts Bar and Guntersville Reservoirs. From Smith and Isom, 1967.

Sampler No.	Date Collected	Hours/Days After Treatment	Material	Mg/l[1] or Mg/Kg BEE	Station[2]	Species
			Watts Bar Reservoir			
19	Prestudy	Control	Fish	<0.14	Control	*Lepomis macrochirus*
25	3-20-66	24 hours	Watermilfoil	<0.14	Gordon Branch	*Myriophyllum spicatum*
18	3-20-66	24 hours	Watermilfoil	8.26	do.	*Myriophyllum spicatum*
16	3-20-66	24 hours	Watermilfoil	3.36	do.	*Myriophyllum spicatum*
21	3-22-66	72 hours	Fish	<0.14	do.	*Lepomis macrochirus*
26	3-23-66	96 hours	Mud	56.0	do.	—
27	3-23-66	96 hours	Mussel	0.38	do.	*Assorted mussels*
22	3-23-66	96 hours	Mud	2.8	do.	—
23	3-23-66	96 hours	Mud	0.95	do.	—
15	3-23-66	96 hours	Fish	<0.14	do.	*Lepomis macrochirus*
8	3-23-66	96 hours	Mussel	0.70	do.	*Elliptio crassidens*
32	4-13-66	24 days	Mud	35.0	do.	—
34	4-13-66	24 days	Mud	0.15	do.	—
33	5-24-66	35 days	Mud	0.14	do.	—
6	5-25-66	50 days	Fish	<0.14	do.	*Lepomis macrochirus*
10	5-26-66	50 days	Fish	<0.14	do.	*Lepomis macrochirus*
48	5-25-66	50 days	Fish	<0.14	do.	*Ictalurus punctatus*
49	5-25-66	50 days	Fish	<0.14	do.	*Silzostedion canadense*
50	5-25-66	50 days	Fish	0.15	do.	*Lepomis macrochirus*
47	1-17-67	10 months	Fish	<0.14	do.	*Pomolobus chrysochloris*
43	1-17-67	10 months	Mud	0.24	do.	—
44	1-17-67	10 months	Mud	0.91	do.	—
45	1-17-67	10 months	Mud	0.28	do.	—
46	1-17-67	10 months	Mud	58.8	do.	—
			Guntersville Reservoir			
17	Prestudy	Control	Mussel	<0.14	Control	*Elliptio crassidens*
2	Prestudy	Control	Asiatic Clams	<0.14	—	*Corbicula manillensis*
5	May 1966	Control	Mud	0.14	—	—
11	4-06-66	24 hours	Mussel	0.25	Ia-1	*Elliptio crassidens*
13	4-06-66	24 hours	Mussel	0.24	"Control"	*Elliptio crassidens*
14	4-06-66	24 hours	Mussel	<0.14	Out-1	*Elliptio crassidens*
30	4-06-66	24 hours	Mussel	1.12	In-3	*Elliptio crassidens*
12	4-06-66	72 hours	Mussel	0.18	"Control"	*Elliptio crassidens*
1	4-08-66	72 hours	Mussel	0.30	In-1	*Elliptio crassidens*
40	4-08-66	72 hours	Mussel	0.98	In-3	*Elliptio crassidens*
41	4-08-66	72 hours	Mussel	1.0	In-2	*Elliptio crassidens*
4	4-11-66	144 hours	Mussel	<0.14	In-1	*Elliptio crassidens*
24	4-11-66	144 hours	Mussel	<0.14	In-3	*Elliptio crassidens*
39	4-21-66	15 days	Fish	<0.14	In-2	*Ictalurus furcatus*
28	4-11-66	144 hours	Mussel	<0.14	Out-3	*Elliptio crassidens*
20	4-11-66	144 hours	Fish	<0.14	Out-1	*Lepomis macrochirus*
7	5-17-66	42 days	Mud	<0.14	Out-1	—
29	5-17-66	42 days	Mud	33.6	In-3	—
31	5-17-66	42 days	Mud	0.14	Out-2	—
3	5-17-66	42 days	Mussel	<0.14	In-1	*Elliptio crassidens*
9	5-17-66	42 days	Mussel	0.20	In-2	*Elliptio crassidens*
35	1-20-67	9 months	Fish	<0.14	In-2	*Dorosoma cepedianum*
36	1-20-67	9 months	Mud	0.34	In-2	—
37	1-20-67	9 months	Mud	0.30	Out-2	—
38	1-20-67	9 months	Mud	0.49	In-3	—
42	1-20-67	9 months	Mud	0.30	"Control"	—

[1] The C.W. England Laboratories converted 2,4-D and its esters to the methyl ester for reporting to TVA; however, for comparison with published data on toxicity (2), all data were converted to the BEE equivalent.

2,4,D — 2,4-dichlorophenoxyacetic acid

________________________________, butoxyethanol ester

________________________________, methyl ester

[2] In = Inside embayment
Out = Outside in river channel external to embayment

Note: The station labelled "Control" received an unplanned application of 2,4-D and, therefore, cannot be a control.

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2,6-dichlorobenzonitrile
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Table 16.

Physical, chemical and toxicological properties of some aquatic herbicides. From Newbold, 1975.

Herbicide	Chemical formula	Mode of action	Persistence in muds	Break-down in water	Form	Treat-ment level (mg/l)	Levels of toxicity to fish LC_{50}	Levels of toxicity to Daphnia LC_{50}
Diquat	1,1'-ethylene-2,2'. dipyridylium dibromide	contact translocation	6 months-1 year	8 to 11 days	Liquid	1.0	90 mg/l, 24 h *Salmo gairdnerii* (Richardson) (Alabaster, 1969)	7.1mg/l, 24 h *Daphnia magna* (Straus) (Crosby & Tucker, 1966)
Paraquat	1,1'-dimethyl-4,4'-diphridylium dichloride	contact translocation	>2 years	7 to 14 days	Liquid	1.0	840 mg/l, 24 h *Rasbora heteromorpha* (Duncker) (Alabaster, 1969)	3.7 mg/l, 48 h *Daphnia pulex* (De Geer) (Sanders & Cope, 1966)
Dichlo-benil	2,6-dichlorobenzo-nitrile	translocation	6 months	2 to 3 months	Granule	1.0-3.0	1.6 mg/l, 10 days *Rutilus rutilus* (L) (Tooby, 1972)	3.7 mg/l, 48 h *Daphnia pulex* (De Geer) (F.W.P.C.A., 1968)
Chlor-*thiamid*	2,6-dichlorothio-*benzamide*	translocation	6 months	2 to 3 *months*	Granule	1.0-*3.0*	41 mg/l, 24 h Ras Bora *heteromorpha* (Duncker) (Alabaster, 1969)	
Terbutryne	2-*tert*-butylamino-4-ethylamino-6-methyl-thio-1,3,5-triazine	disrupts photo-synthesis	Not known, likely to be very persistent	>3 months	Granule	0.025-0.1	3.5 mg/l, 96 h *Salmo gairdnerii* (Richardson) (Tyson, 1974)	1.4 mg/l, 48 h *Daphnia magna* (Straus) (Tyson, 1974)

Residues in Water and Soil. Diquat is inactivated by contact with soil particles, so it cannot be used in turbid conditions. Newbold (1975) reported complete degradation of 1.0 ppm in water in 8 to 11 days. Persistence in bottom muds lasts from six months to one year (Table 16). Once the material is tightly bound to hydrosoil it is, in effect, out of the system and remains bound to soil particles. In this condition, it cannot be re-released into the water column nor can it be absorbed and re-enter the ecosystem via plant uptake. Diquat is degraded in hydrosoil microbially.

Mackenzie (1969) reported three residue trials conducted in southern Florida in 1966. Diquat was applied to Florida elodea (hydrilla) at the rate of 0.5 ppm. Water for residue analysis was collected at the surface, mid-depth and bottom, and plant hydrosoil samples were collected 1 day, 8 days, 2 weeks, and 3 weeks after treatment.

Data in Tables 17, 18, and 19 indicate that except for Trial 1, diquat residues were not found 14 days after treatment. Apparently, submersed aquatic weeds rapidly absorb diquat soon after treatment. Rapid uptake by plants is evidenced; water residue values averaged 0.22 ppm 1 day after treatment with 0.5 ppm diquat. Laboratory studies have further demonstrated rapid absorption of diquat by hydrilla (Sutton et al. 1972).

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Table 17.
Details of residue trials 1, 2, and 3, Dade County, Florida, 1966. From Mackenzie, 1969.

Residue Trial number	Canal Dimensions Length	Canal Dimensions Width	Canal Dimensions Depth	Density of Elodea[1]	Location	Previous Diquat History
	feet	feet	feet			
1	2120	85	13	30	Andover "A"	No diquat
2	1500	45	6	75	Carol City "A8"	No diquat
3	1720	45	12	100	Hefler Homes "N"	Treated two times in 1965 without success

[1] Expressed as percentage of canal volume occupied by elodea.

Table 18.
Water samples - diquat residues as related to time after treatment and depth in residue trials 1, 2 and 3. From Mackenzie, 1969.

Sample level in water profile	Residue Trial No.	Time After Treatment 1 day	8 days	2 weeks	3 weeks
		ppmw	ppmw	ppmw	ppmw
Top foot of water	1	0.28	0.10	0.06	0.00
	2	0.20	0.04	0.00	0.00
	3	0.24	0.00	0.00	0.00
Mid section of profile[1]	1	0.09	0.10	0.09	0.00
	2	0.26	0.03	0.00	0.00
	3	0.45	0.00	0.00	0.00
Bottom foot of water	1	0.27	0.11	0.07	0.00
	2	0.22	0.08	0.00	0.00
	3	0.01	0.01	0.00	0.00

[1] 1 and 3—6 ft. below surface
2—3 ft. feet below surface.

Table 19.
Diquat residues in elodea based on dry weight as related to time after treatment in residue trials 1, 2 and 3. From Mackenzie, 1969.

Sample site	Time after Treatment 1 day	8 days	2 weeks	3 weeks
	ppmw	ppmw	ppmw	ppmw
Residue Trial 1 - treated area	9.08	11.55	Elodea eradicated	Elodea eradicated
Residue Trial 2 - treated area	6.58	9.23	6.63	4.90
Residue Trial 3 - treated area	0.17	0.06	0.04	0.04
Residue Trial 3 - area affected by flow from treated area	—	—	0.29	—

Mackenzie's (1969) field residue trials also determined diquat content of submersed weeds (Table 20) on a mg diquat/kg dry weight basis. One day after treatment, hydrilla contained 9.08 ppm diquat in Trial 1 and 6.58 ppm in Trial 2, but in Trial 3, which was affected by water flow, the plants only contained 0.17 ppm diquat 24 hours after treatment.

Table 20.
Bottom soil samples - diquat residues based on dry weight as related to time after treatment in residue trials 1, 2 and 3. From Mackenzie, 1969.

Residue trial number	Time after treatment			
	1 day	8 days	2 weeks	3 weeks
	ppmw	ppmw	ppmw	ppmw
1	0.33	3.52	6.84	3.98
2	0.85	3.68	2.60	3.14
3	0.00	0.00	1.74	0.00

Bottom mud samples indicated that diquat persisted in hydrosoil longer than in any other portion of the ecosystem. The data are variable and additional samples should have been taken, but Mackenzie's hydrosoil residue data suggest maximum diquat accumulation in hydrosoil 2 weeks after treatment (mean of his trials was 3.71 ppm).

Interestingly, this was the same length of time generally required for submersed weeds to disintegrate and portions of them to sink to the hydrosoil. Visual evaluations of the residue plots (Table 21) 1 and 2 weeks after treatment showed that the greatest increase in plant control was obtained during this time. This suggests that submersed weeds, soon after treatment and in the process of sinking, carry diquat to the hydrosoil.

Table 21.
Florida elodea control - results in residue trials 1, 2 and 3 treated with 0.5 ppmw diquat, 1966. From Mackenzie, 1969.

Residue trial number	Infestion of Elodea	Deposits on Elodea: Calcium carbonate	Deposits on Elodea: Algae	Control: 1 wk.	2 wks.	3 wks.	6 wks.	8 wks.
	%			%	%	%	%	%
1	30 - marginal	None	Medium	75	99	100	100	100
2	75 - center open	Medium	Heavy	70	85	95	95	85[1]
3	100	Medium	Heavy	35	65	90	80	70[2]

[1] Retreatment with 0.5 ppmw diquat at 4 months gave 100% control for the remainder of 1966.
[2] Two retreatments at 0.5 ppmw diquat did not give satisfactory control in 1966.

In summary, diquat applied at 0.5 to 1.0 ppm is widely used to control submersed aquatic plants. Data indicate that plants have absorbed more than 50% of the diquat within hours after treatment. Suspended soils, hydrosoil, and breakdown by ultraviolet radiation all contribute to removal of the chemical from the water (Simsiman et al. 1976).

Endothall [7-oxabicyclo(2.2.1)heptane-2,3-dicarboxylic acid)]

Endothall, as an organic acid, has the potential of being manufactured in many salt, amine, or other formulations. The formulations most widely used in aquatic weed control are the potassium salt and the dimethyl-alkylamine salt of endothall.

Endothall is used primarily to control submersed aquatic weeds and is applied at rates from 1 to 2 ppm, depending on the plant species to be controlled and on formulation. It is injected either as liquid into water or as sinking pellets applied to the surface.

Residues in Water and Soil. Hiltibran (1962) conducted some of the earliest published residue research concerning endothall. Endothall was approved for aquatic weed control in 1960, and is one of the older, most extensively studied organic aquatic herbicides.

Hiltibran's (1962) studies compared several combinations of tap water, lake water, fish, mud, and plant debris placed in aquariums on degradation of endothall. There were considerable differences in degradation among treatments (Table 22). His data indicate that microcosms that included variations of lake water, mud, plant debris, and fish have the most rapid degradation of the disodium salt of endothall.

Further field studies were conducted in plastic enclosures placed in farm ponds (Table 23) where there were less variation and more rapid degradation after application of 1 ppm endothall; none was detectable an average of 36 hours after treatment. None was detectable 66 hours after application of 5 ppm, and none was detectable 72 hours after application of 10 ppm.

A more recent study and review of endothall residues has been compiled by Sikka and Rice (1973). These authors discussed variation in the rate of degradation of the various salts of endothall, and particularly the disappearance of endothall from water, accompanied by smaller but concurrent increases in endothall concentrations in hydrosoil.

Dipotassium endothall (Aquathol^R K) was applied to a 0.1 acre nonflowing pond (4 ft deep) at a rate of 2 ppm. Within the first 3 days after treatment, the endothall concentration was decreased by 55% in the water (Figure 30). As the endothall concentration in the water decreased between days 4 and 22 after treatment, there was a slight increase in hydrosoil endothall content. After 36 days, endothall could not be detected in the water but persisted in the hydrosoil, becoming non-detectable 44 days after treatment.

Table 22.
Disappearance of endothall from laboratory aquaria containing various combinations of tap water, lake water, 4 to 6 live fish, plant debris, and mud. From Hiltibran, 1962.

Endothal applied ppm	Tank contents	Hours to reach 0.5 ppm endothal	Days to reach 0.1 ppm endothal
1	Lake water,[a] mud, fish	98	8
1	Lake water, fish	98	8
1	Tap water, fish, mud	98	8
1	Tap water, fish	265	21
1	Tap water, 1/2, lake water, 1/2; fish, mud	98	8
1	Lake water[b]	403	—
1	Lake water	32	—
1	Lake water, fish	32	—
1	Lake water, fish, mud	32	—
1	Lake water, fish, mud	32	—
5	Lake water	957	61
5	Lake water	478	40
5	Lake water, mud	236	13
5	Lake water, mud	236	13
5	Lake water	146	9
5	Lake water, fish	—	5[c]
5	Lake water, fish	166	6[c]
5	Tap water, mud	166	6[c]
5	Tap water, 1/2, lake water, 1/2; fish, mud	166	6[c]
5	Lake water, fish, mud	166	6[c]
5	Lake water, plant debris	172	12
5	Tap water, fish	502	21
10	Lake water, plant debris	364	22
10	Lake water, mud	364	22
10	Lake water, plant debris, mud	364	22
10	Tap water, mud	337	20
10	Tap water, plant debris	337	20
10	Tap water, plant debris, mud	337	20
10	Lake water, plant debris	172	12

[a] Lake water from Arrowhead Pond unless otherwise specified.
[b] Lake water for this tank was from Allerton Lake.
[c] Concentration of endothal was 0.3 ppm instead of 0.1 ppm.

Table 23.
Disappearance of endothall from plastic-enclosed test plots of aquatic vegetation in farm ponds. From Hiltibran, 1962.

Pond	Plot size, feet	Rate ppm	Time since application, hours	Conc. ppm	Hours until endothal was not detected
Arrowhead	35 x 8	1	2.0	1.0	24
Hay's	15 x 15	1	0.5	1.7	24
Sage's	20 x 5	1	0.5	1.0	48
Hay's	10 x 10	1	0.5	1.0	48
Arrowhead	30 x 8	5	24.0	0.8	48
Hays'	15 x 15	5	24.0	1.7	48
Sage's	10 x 10	5	48.0	0.8	72
Hays'	10 x 10	5	48.0	0.8	96
Arrowhead	15 x 15	10	24.0	0.8	48
Hays'	10 x 10	10	168.0	1.0	—
Sage's	10 x 10	10	48. 0	1.0	72
Hays'	10 x 10	10	72.0	1.0	96

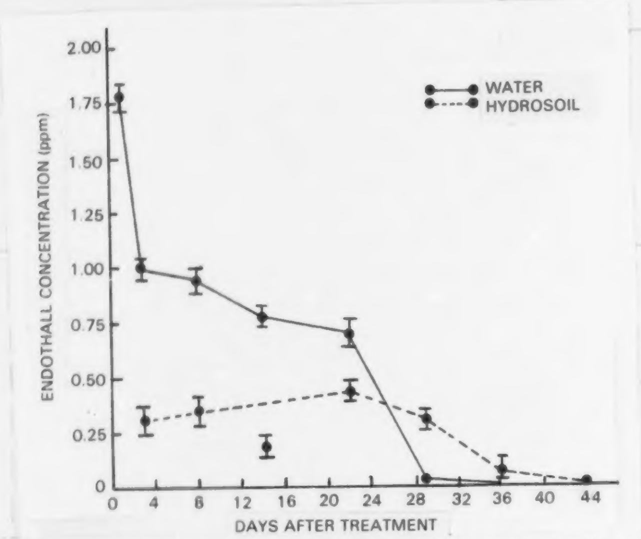

Figure 30. Endothall residues in water and the top 1-inch of hydrosoil of a treated farm pond, with time. The bars represent the range of duplicate values. From Sikka and Rice, 1973.

In most studies, degradation of herbicide occurred most rapidly in the field. However, aquaria studies by Sikka and Rice (1973) indicated more rapid degradation in aquaria than in the pond study. One day after treatment with 2 and 4 ppm dipotassium endothall, 80 and 86% of the residue, respectively, had disappeared from the water (Figure 31). Endothall was not detected seven days after treatment, compared to 36 days in field studies. Biodegradation of endothall on hydrosoil also occurred more rapidly in aquaria. Treatment with 2 ppm resulted in non-detectable hydrosoil endothall at day 22, and rates of 4 ppm were below detection at day 35 after treatment.

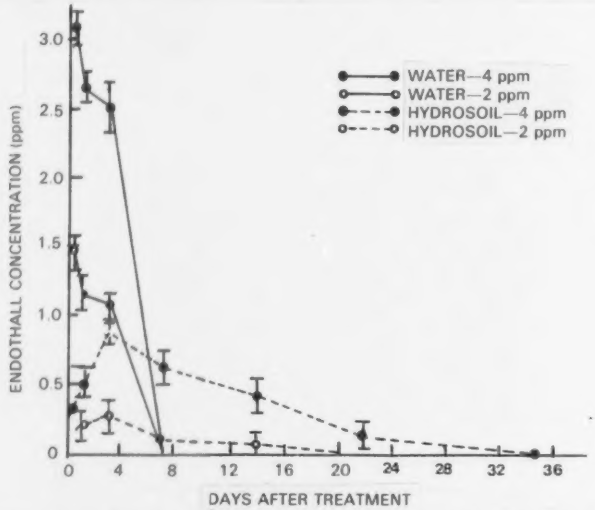

Figure 31. Endothall residues in water and hydrosoil of aquaria treated with 2 and 4 ppm of the herbicide. The bars represent the range of duplicate values. From Sikka and Rice, 1973.

As with the other submersed aquatic weed herbicides, endothall is absorbed rapidly after application by plants, organic matter, and hydrosoil. Residues in water drop at least 50% after 24 hours and gradually decrease to non-detectable between two and three weeks after treatment. Endothall persists slightly longer in hydrosoil, but hydrosoil concentrations rarely exceed an order of magnitude below treatment levels.

Copper Salts and Organic Copper Complexes

Copper sulfate (copper sulfate pentahydrate;CSP;$CuSO_4.5H_2O$) has been used for over 100 years as a fungicide and algicide. Before the development of organic herbicides in the 1940s, CSP and other mineral salts were applied to soil as soil sterilants and were sprayed on foliage of floating aquatic plants. Copper compounds were, however, most effective in the aquatic environment as algicides.

Copper is still widely used as an algicide, but this discussion will be limited to discussion of copper for control of submersed vascular aquatic plants. When metallic copper (usually CSP) was used to control vascular submersed aquatic weeds, such as milfoil, Elodea, and hydrilla, it was soon discovered that concentrations required to control aquatic plants (4-8 ppm) were very close to the threshold of fish toxicity. Consequently, fish kills often occurred when CSP was used for aquatic weed control.

In the mid 1960s, chelated organic copper complexes were developed and greatly reduced the toxicity and corrosiveness of copper, principally by formulating the sulfate (SO_4^-) radical out of the mixture (Table 24). Concurrently, research was conducted with combinations of diqut and copper compounds. During these studies (Mackenzie and Hall 1967; Sutton et al. 1972; and Haller and Sutton 1973), it was discovered that copper-diquat and copper-endothall mixtures had a synergistic effect on aquatic weeds, particularly hydrilla. The use of 4 lb diquat and 2 to 4 lb copper per acre provided good (albeit expensive) submersed weed control at herbicide concentrations well below the level of fish toxicity.

Liquid formulations of copper organochelate formulations have almost entirely replaced CSP. Copper complexes are generally injected into submersed weed mats in combination with diquat or endothall. There are several copper chelate products on the market. The three principal products are: copper triethanolamine, copper ethylene diamine, and copper diethylene triamine.

Residues in Water and Soil. Major large-scale field tests of diquat and copper combinations were conducted in 1971 by the U.S. Army Corps of Engineers in cooperation with several state and federal agencies (Mobley et al. 1971). Herbicide residues, invertebrate populations, water quality investigations, etc., were monitored concurrently in several stations in the 250-acre treatment site. The herbicides diquat and Cutrine^R (copper triethanolamine) were applied in combination at rates of 0.75 ppm diquat and 0.14 ppm copper. The area treated was 85 acres, which comprised 694 acre feet of water.

Table 24.
Toxicity of commonly used aquatic herbicides to bluegill fingerlings.
Mortality is cumulative across the table. From Haller, unpublished data.

	Percent mortality			
n=20	2 hr	20 hr	72 hr	96 hr
Diquat:				
0.5	0	10	10	10
1.0	0	0	0	0
2.5	0	10	10	10
5.0	0	5	5	5
10.0	0	5	10	10
20.0	0	5	30	35
Cutrine Plus:				
5.0	0	0	0	0
10.0	0	5	5	5
20.0	10	100	100	100
40.0	40	100	100	100
80.0	5	100	100	100
Copper sulfate:				
5.0	0	0	0	0
10.0	60	100	100	100
20.0	100	100	100	100
40.0	100	100	100	100
Weedar 64: (2,4-D)				
5.0	0	5	5	5
10.0	0	0	0	0
20.0	0	0	0	0
40.0	0	15	15	15
80.0	0	100	100	100
Control:				
A	0	0	0	0
B	0	0	0	0

In an adjacent area (165 acres), diquat (1.0 ppm) and 0.8 ppm of copper as copper sulfate were applied to nearly 1,400 acre feet of water.

The total area treated was 250 acres in a several thousand acre impoundment. The site was a hydrilla-infested embayment on the north side of a reservoir. Weed control was achieved in the diquat-copper sulfate treatment area, but minimal vegetation control occurred in the diquat-Cutrine site due to the lower total herbicide concentrations used and internal water exchange in the reservoir.

Copper residue data from the water are presented in Table 25. The background, or pretreatment water copper, varied from 0.001 to 0.011 ppm Cu. Fourteen days after treatment, a range of 0.005 to 0.014 ppm copper was found. During this period, it appeared that the aquatic plants had absorbed most of the copper from the water column. The information presented in Table 26 indicates very high copper concentrations in the hydrilla plants almost immediately after chemical treatment.

add: (T= top, B= bottom) after caption before period.

Table 25.

Copper in solution after treatment of Inglis Reservoir. Samples for stations 1 to 4 are from the area treated with diquat plus cutrine and stations 5 to 8 from the diquat plus CSP (T = top, B = bottom). From Mobley et al., 1971.

Station Number	Water Depth	Copper in solution (ppmw)				
		Days after treatment				
		0	1	3	7	14
1	T	.004	.021	.014	.012	.007
	B	.009	.036	.016	.012	.007
2	T	.010	.010	.012	.012	.005
	B	.006	.024	.014	.012	.007
3	T	.009	.125	.024	.007	.005
	B	.002	.048	.018	.015	.006
4	T	.005	.005	.011	.006	.006
	B	.002	.018	.010	.006	.005
5	T	.009	.060	.027	.012	.008
	B	.002	.165	.026	.019	.013
6	T	.011	.038	.024	.006	.007
	B	.005	.034	.026	.007	.010
7	T	.004	.350	.080	.012	.008
	B	.005	.070	.083	.019	.012
8	T	.006	.215	.125	.012	.014
	B	.001	.100	.025	.007	.011

add as above before period (T= top, B= bottom).

Table 26.

Copper content of hydrilla after treatment of Inglis Reservoir. Plant samples for stations 1 to 4 are from the area treated with diquat plus cutrine and stations 5 to 8 from the diquat plus CSP (T = top, B = bottom). From Mobley et al., 1971.

Station Number	Water Depth	Copper in dry plant material (ppmw)					
		Days after treatment					
		0	1	3	7	14	28
1	T	4	252	75	109	32	23
	B	7	31	22	24	40	12
2	T	6	258	104	172	34	11
	B	10	17	14	18	18	9
3	T	4	260	550	43	92	15
	B	7	15	15	20	42	22
4	T	4	196	103	52	38	12
	B	3	16	35	11	17	9
5	T	7	1,870	1,470	2,000	218	[a]
	B	7	38	90	79	51	36
6	T	7	126	214	75	50	11
	B	12	12	16	11	10	14
7	T	7	1,040	1,750	1,750	615	340
	B	6	43	112	1,480	3,240	410
8	T	6	2,460	1,230	990	550	49
	B	4	1,050	294	238	1,580	32

[a] No sample available.

The large-scale test indicated that in a heavily vegetated area, diquat was essentially nondetectable in the water seven days after treatment.

Several factors account for disappearance of copper ion from the aquatic ecosystem. In tests such as this, copper is distributed over time into the rest of the waterbody and is absorbed and diluted by algae and aquatic plants. In certain waters, notably hard waters, copper ions are precipitated by carbonate ions and are thus incorporated into sediments.

Since the 1970 field test, copper and diquat rates have been reduced to those currently recommended: 4-6 lb diquat and 4-8 lb copper per acre, independent of depth. In water six feet deep, these rates provide a concentration of 0.4 ppm diquat and 0.5 ppm of copper.

In summary, the hydrosoil is the primary sump when copper treatments are conducted in closed ponds and non-flowing situations. In large reservoirs, lakes, canals and rivers, none of which have ever been treated in total, the copper residue is diluted in water, plants and hydrosoil where it cycles in the ecosystem at near background concentrations.

Dalapon (2,2-dichloropropionic acid)

Dalapon is a selective herbicide which is most effective on grasses and other monocots. As an acid, it can be formulated into several salts and other compounds. The vast majority of dalapon is formulated as mixtures of sodium and magnesium salts.

Dalapon has a very low toxicity to animals (Paynter et al. 1960) and is safe for fish and invertebrates above 100 ppm. In aquatic systems it is used for emergent weed control only, principally cattail and other ditchbank species. It is often used in combination with 2,4-D in ditchbank weed control. Dalapon kills or suppresses grass species and the 2,4-D formulation controls the broadleaf (dicot) weed species.

As a ditchbank herbicide, very little dalapon is expected to be found in water. Thus, dalapon's occurrence in water would be incidental, and significant concentrations are unlikely to occur in the water or soil of lakes, reservoirs, or rivers.

Residues in Water and Soil. It is likely that dalapon residues occur in small agricultural ditches and along canals where cattail and grasses have been sprayed. As a simple organic acid, its breakdown in water and hydrosoil is expected to be rapid.

Dalapon is applied as a foliar spray on ditchbank grasses where it is apparently rapidly absorbed through the leaves and roots and translocated to meristematic regions (Blanchard et al. 1960).

Dalapon is most widely used in terrestrial weed control. Holstun and Loomis (1956) studied leaching and decomposition of dalapon in soils. Solutions of dalapon (120 ppm) were applied to soils with varying moisture, temperature, and organic matter contents. In general, the degradation of dalapon is enhanced by soils which have high moisture, high temperature and high organic matter contents.

The most extensive compilation of the use, toxicity, degradation, and environmental effects of dalapon was published by Kenaga (1973). Kenaga compared the chemical oxygen demand (COD) and biological oxygen demand (BOD) of dalapon in water and found that dalapon is completely broken down through bacterial degradation. The end products of dalapon degradation are CO_2, H_2O, NaCl, and CL_2 (Figure 32).

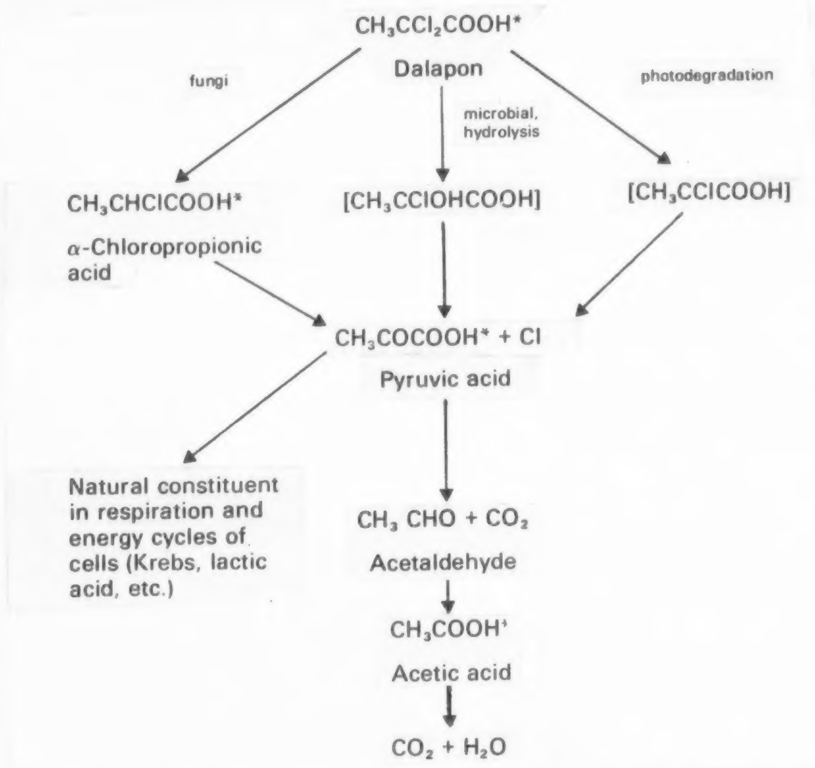
$CH_3CCl_2COOH^*$
Dalapon
fungi
microbial, hydrolysis
photodegradation
$CH_3CHClCOOH^*$
α-Chloropropionic acid
$[CH_3CClOHCOOH]$
$[CH_3CClCOOH]$
$CH_3COCOOH^* + Cl$
Pyruvic acid
Natural constituent in respiration and energy cycles of cells (Krebs, lactic acid, etc.)
$CH_3CHO + CO_2$
Acetaldehyde
CH_3COOH^*
Acetic acid
$CO_2 + H_2O$

Figure 32. Chart of the major metabolic and degradative routes of dalapon: the starred acids will be in equilibrium with their anions with the position of the equilibrium depending on pH and the specific cations in a particular environment and the compounds in brackets will be transient. From Kenaga, 1973.

Frank et al. (1970) applied dalapon at rates of 6.7 to 20 lb (active ingredient)/acre to ditchbank weeds along irrigation canals. Water collected immediately after spraying contained insignificant levels of dalapon (0.02 to 0.37 ppm). It was concluded that dalapon at these levels would be degraded rapidly in the aquatic ecosystem, and even the highest (0.37 ppm) posed no threat to use for irrigating agricultural crops.

The most recent investigation concerning dalapon and its environmental effects on the aquatic ecosystem is that of Brooker (1976). Brooker treated emergent aquatic weeds along canal banks with dalapon and 2,4-D. Dalapon (25 kg a.i./ha) was applied in split application once in the fall of 1974, and again (12 kg a.i./ha) in the following spring with 2,4-D (1 kg a.i./ha). These treatments effectively controlled emergent grasses and broadleaf weeds. The chemically treated canal reach was compared to a similar upstream reach which received hand weeding.

Brooker analyzed 140 water samples to determine the residues of the two chemicals following application. After the fall treatment, dalapon was found in the water at 38 ppb (one hour post treatment). After 8 days, dalapon residues were no longer detectable in the treated area, and only trace amounts (<1ppb) were detectable downstream (2 to 3 km).

Brooker was able to calculate the amount of herbicide leaving the treated area by utilizing flow rate and residues (Table 27). His calculations indicate that only 0.0001% of the fall-applied dalapon left the target area after treatment, and 0.01 and 0.002% of the dalapon and 2,4-D left the treatment area, respectively, after the spring application.

Table 27.
Loss of herbicide from the chemical reach. From Brooker, 1976.

Herbicide application	Mass of herbicide applied to reach (g)	Mass of herbicide leaving the reach (g)	% herbicide leaving the reach
Dalapon (Sept. 1973)	2250	0.002	0.0001
Dalapon (May 1974)	1080	0.006	0.01
2,4-D (May 1974)	90	0.002	0.002

As in other studies, residue of dalapon in canal water of Brooker's study was insignificant. The effects of chemical control of emergent weeds in the canals of Essex, England, generally increased the growth of replacement aquatic plants (Lemna and Callitriche) compared to similar areas of hand weeding.

Effects of Herbicides on Water Quality

Previous sections of this report have discussed herbicide residues in the aquatic environment (flora, water, soil, and fauna). Effects of aquatic herbicides on water quality will be discussed separately in this section because the herbicides per se have no effect on traditional aspects of water quality (physio-chemical parameters). The changes that occur in water quality are not directly a result of herbicide application, but rather are a result of dying vegetation and changing of aquatic succession.

Research studies conducted with the objective of determining effects of weed control on water quality are numerous, and results are as variable as the containers used in the studies: erlenmeyer flasks to 10,000 acre lakes. Water quality changes associated with chemical aquatic weed control result from release of nutrients and organic matter from the decaying weeds. Even then, not all nutrients contained in aquatic vegetation are released to the water, considerable amounts of nutrients are incorporated into the soil and benthic community as the vegetation falls to the hydrosoil (Strange 1976).

In small, enclosed systems such as aquaria, plastic pools, and similar nonflowing, unnatural systems, herbicide application and subsequent decay of aquatic vegetation invariably leads to at least temporary increases in dissolved plant nutrients and algal populations (Hestand and Carter 1978; Walker 1963).

Nutrient released from treatment of peripheral emergent aquatic plants in noneuthrophic lakes or river systems produce no changes or only temporary minor changes in water quality and phytoplankton populations.

Factors that determine whether or not major changes in water chemistry occur after weed control operations are dependent upon the physical and chemical characteristics of a given water body (Table 28).

Table 28.
Characteristics of water which, if combined, will invariably produce major changes in water quality if treated for aquatic weed control.

High water temperature.
High plant biomass to be controlled.
Shallow water, preferably eutrophic.
High percentage of water surface area teated.
Closed ponds, non-flowing situations.

According to Table 28, an algal bloom and deterioration of water quality will invariably occur in the following situation: eutrophic shallow nonflowing pond (1-2 m deep), 90% covered with water hyacinth (200 metric tons fresh wt/acre), sprayed in July.

On the other end of the scale, the following treatment situation would produce no measurable effect on water quality outside of the immediate treatment area: oligotrophic, deep lake (100 acres), 100 feet long, 4 feet wide fringe of shoreline Typha plants, sprayed in early spring at initation of new growth.

These two extreme examples are used to emphasize the variability of potential environmental effects of chemical control of aquatic weeds. A trained and experienced aquatic biologist can accurately predict the outcome of various weed control operations.

EPA and State pesticide laws require that permits be obtained by aquatic weed control firms before control operations are conducted (particularly in public waters). The permitting agency therefore must weigh the benefit-risk ratio before permitting the operation.

Currently, Federal and State laws closely monitor and regulate aquatic herbicide labelling directions and registration. The rigorous and extensive requirements for herbicide registration for aquatic ecosystems insures that only the safest possible chemicals are used in aquatic weed control. Any detrimental environmental effects that occur as a result of chemical control of aquatic weeds are not the fault of the chemical, but rather the reduction and decay of the controlled vegetation.

Obviously, all alternative means of aquatic weed control must be evaluated before wise management decisions can be made. All means (mechanical, biological, chemical, drawdowns, etc.) of aquatic weed control have environmental impact, if for no other reason than eliminating undesirable vegetation.

Better aquatic weed control means control of aquatic plants more economically and without increasing the environmental impact of control operations.

Effects of Herbicides on Aquatic Organisms

The herbicide 2,4-D is used extensively for the control of aquatic plants. Several formulations of this chemical are used. For example, the diethylamine salt is used for water hyacinth control, whereas 2,4-D BEE is used for milfoil. The chemical formulation often determines the toxicity of the chemical to aquatic life. Hiltibran (1967) pointed out that the water soluble derivatives of 2,4-D were less toxic than their ester derivatives, and Walker (1964a,b) found that butyl, butoxyethanol and propylene glycol butyl ether esters more toxic to fish than several other 2,4-D formulations. The diethylamine salt of 2,4-D is the derivative used most often for hyacinth control and appears to be the least toxic of the forms.

2,4-D is usually applied at a rate of 2 mg/l to the treatment area. It is possible, however, to have areas of higher concentration, especially where plants are thick and higher rates are applied. All formulations [2,4-D PGBE; 2,4-D BEE; 2,4-D IOE and 2,4-D (diethylamine salt)] are relatively nontoxic to aquatic animals. Acute toxicity levels (LC50), as tabulated in Table 29 (Anon. 1973), indicate that LC50 values range from 100 g/l (2,4-DPGBE) to having no effect at 100,000 g/lN 48 hours. Other studies have reported insignificant changes in the abundance or distribution of freshwater plankton or benthos (Pierce 1968, 1969). Alabaster (1969) reported an LC50 of 250 mg/l (24 hours) for rainbow trout, *Salmo gairdnerii*. Hughes and Davis (1962) found 48-hour TL_m values for bluegills (*Lepomis microchirus*) to range from 0.8 to 840 mg/l acid equivalent for several formulations of 2,4-D. McCorkle et al (1977) considered 2,4-D acid and diethylamine salts of the compound to be safe for use around catfish ponds as few mortalities occurred when channel catfish (*Ictalurus punctatus*) were exposed to the chemical. Application of 25 to 30 l 2,4-D/acre did not affect fish in hatchery ponds (Mackenthun 1961). In most of the studies reported above, larger fish were used. Hiltibran (1967) utilize small bluegill and fry of the fish in his study (Table 30) and found 2,4-D to be relatively nontoxic to these organisms. The diemethylamine salt had no effect as all test fish held to term. Mount and Stephen (1967) reported that fathead minnows (*Pimephales promelas*) reproduced normally when exposed to 0.3 ppm 2,4-D.

The effects of 2,4-D on marine organisms has also been investigated. Rawls (1965) tested eastern oysters, soft shell clams, and various fish species in conjunction with herbicidal treatments. Chemicals were applied as follows: 2,4-D acetamide 20 lb/acre; polypropylene glycol butyl ether esters of 2,4-D 20 lb/acre; butoxy ethanol ester of 2,4-D 20 to 120 lb/acre; isoctyl ester of 2,4-D 20 to 60 lb/acre. He concluded, after a one month trial, that only 2,4-D acetamide appeared dangerously toxic. Butler (1965) found pink shrimp (*Penaeus duorarum*) showed no effect at 1.0 ppm 2,4-D at 48 hours, whereas brown shrimp (*Penaeus aztecus*) showed 10% mortality or paralysis at 2.0 ppm at 48 hours.

Residue studies indicate similar findings as do toxicity tests. Fish residue studies indicate little 2,4-D uptake. Schultz and Whitney (1974) determined residues in 60 fish samples after water hyacinths were sprayed with the dodecyl-tetradecyl amine salts of 2,4-D (DTA-2,4-D) at a rate of 4.48 kg acid equivalent/ha. The initial treatment was followed with spot treatments of DTA-2,4-D or the diemethylaine salt of 2,4-D (DMA-2,4-d), or both, over a four-month period. Residues ranged from zero in 40 samples to greater than 0.1 mg/kg in two samples (Table 31). Cope et al. (1970) found no residues in whole bluegills with pond treatments of 0.5 ppm or less with propylene glycol butyl ether esters of 2,4-D.

Table 29.
Herbicides, fungicides, defoliants. From Anon., 1973.

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No affect µg/liter	Reference
		µg/liter	hours			
ACROLEIN AQUALIN	FISH					
	Lepomis macrochirus	80	24			Bond et al. 1960
	Salmo trutta	46	24			Burdick et al. 1964
	Lepomis macrochirus	79	24			"
AMINOTRIAZOLE AMITROL	CRUSTACEAN					
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970
	Daphnia magna	30000	48			"
	Cypridopsis vidua	32000	48			"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Lepomis macrochirus				100,000 µg/l 48 hr.	Sanders 1970
	Oncorhvncus kisutch	325000	48			Bond et al. 1960
BALAN	CRUSTACEAN					
	Gammarus faciatus	1100	96			Sanders 1970
BENSULFIDE	CRUSTACEAN					
	Gammarus faciatus	1400	96			Sanders 1970
CHLOROXURON	FISH					
	Lepom s macrochirus	25000	48			Hughes and Davis 1964
CIPC	FISH					
	Lepomis macrochirus	8000	48			Hughes and Davis 1964
DACTHAL	FISH					
	Lepomis macrochirus	700000	48			Hughes and Davis 1964
DALAPON (SODIUM SALT)	CRUSTACEAN					
	Simocephalus serrulatus	16000	48			Sanders and Cope 1966
	Daphnia pulex	11000	48			"
	INSECT					
	Pteronarcys californica				100,000 µg/l 96 hr.	Sanders and Cope 1968
	FISH					
	Pimephales promelas	290000	96			Surber and Pickering 1962
	Lepomis macrochirus	290000	96			"
	Oncorhynchus kisutch	340000	48			Bond et al 1960
DEF	CRUSTACEAN					
	Gammarus lacustris	100	96			Sander- 1969
	INSECT					
	Pteronarcys californica	2100	96			Sanders and Cope 1968
DEXON	CRUSTACEAN					
	Gammarus lacustris	3700	96			Sanders 1969
	INSECT					
	Pteronarcys californica	24000	96			Sanders and Cope 1968
DICAMBA	CRUSTACEAN					
	Gammarus lacustris	3900	96			Sanders 1969
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970
	Daphnia magna				100,000 µg/l 48 hr.	"
	Cypridopsis vidua				100,000 µg/l 48 hr.	"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Lepomis macrochirus	20000	48			Hughes and Davis 1962
DICHLOBENIL CASARON®	CRUSTACEAN					
	Gammarus lacustris	11000	96			Sanders 1969
	Gammarus fasciatus	10000	96			Sanders 1970
	Hyallella azteca	8500	96			Wilson and Bond 1969
	Simocephalus serrulatus	5800	48			Sanders and Cope 1968
	Daphnia pulex	3700	48			Sanders and Cope 1968
	Daphnia pulex	3700	48			"
	Daphnia magna	10000	48			Sanders 1970
	Cypridopsis vidua	7800	48			"
	Asellus brevicaudus	34000	48			"
	Palaemonetes kadiakensis	9000	48			"
	Orconectes nais	22000	48			"
	INSECTS					
	Pteronarcys californica	7000	96			Sanders and Cope 1968
	Tendipedidae	7800	96			Wilson and Bond 1969
	Callibaetes sp.	10300	96			"
	Limnephilus	13000	96			"
	Enallegma	20700	96			"
	FISH					
	Lepomis macrochirus	20000	48			

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Table 29.
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Pesticide	Organism	Acute toxicity LC50 μg/liter	Acute toxicity LC50 hours	Sub-acute effects μg/liter	No effect μg/liter	Reference
DICHLONE PHYGON XL	CRUSTACEAN					
	Gammarus lacustris	1100	96			Sanders 1969
	Gammarus fasciatus	100	96			Sanders 1970
	Daphnia magna	25	48			"
	Cypridopsis vidua	120	48			"
	Asselus brevicaudus	200	48			"
	Palaemonetes kadiakensis	450	48			"
	Orconectes nais	3200	48			"
	FISH					
	Lepomis macrochirus	70	48			Bond et al. 1960
	Micropterus salmoides	120	48			Hughes and Davis 1962
DIQUAT	CRUSTACEAN					
	Hyallella azteca	48	96			Wilson and Bond 1969
	INSECTS					
	Callibaetes sp.	16400	96			Wilson and Bond 1969
	Limnephilus	33000	96			"
	Tendipedidae	>100000	96			"
	Enallagma	>100000	96			"
	FISH					
	Pimephales promelas	14000	96			Surber and Pickering 1962
	Lepomis macrochirus	35000	96			Gilderhus 1967
	Micropterus salmoides	7800	96			Surber and Pickering 1962
	Esox lucius	16000	48			Gilderhus 1967
	Stizostedion vitreum vitreum	2100	96			"
	Salmo gairdneri	11200	48			"
	Oncorhynchus tshawytscha	28500	48			Bond et al. 1960
DIURON	CRUSTACEAN					
	Gammarus lacustris	160	96			Sanders 1969
	Gammarus fasciatus	700	96			Sanders 1970
	Simocephalus serrulatus	2000	48			Sanders and Cope 1966
	Daphnia pulex	1400	48			"
	INSECT					
	Pteronarcys californica	1200	96			Sanders and Cope 1968
	FISH					
	Oncorhynchus kisutch	16000	48			Bond et al. 1960
DIFOLITAN	CRUSTACEAN					
	Gammarus lacustris	800	96			Sanders 1969
	INSECT					
	Pteronarcys californica	40	96			Sanders and Cope 1968
DINITROBUTYL PHENOL	CRUSTACEAN					
	Gammarus fasciatus	1800	96			Sanders 1970
DIPHENAMID	CRUSTACEAN					
	Gammarus fasciatus				100,000 μg/l 48 hr.	Sanders 1970
	Daphnia magna	56000	48			"
	Cypridopsis vidua	50000	48			"
	Asellus brevicaudus				100,000 μg/l 48 hr.	"
	Palaemonetes kadiakensis	58000	48			"
	Orconectes nais				100,000 μg/l 48 hr.	"
DURSBAN	CRUSTACEAN					
	Gammarus lacustris	0.11	96			Sanders 1969
	INSECT					
	Pteronarcys californica	10	96			Sanders and Cope 1968
	Pteronarcella badia	0.38	96			"
	Claassenia sabulosa	0.57	96			"
2,4-D (PGBE)	CRUSTACEAN					
	Gammarus lacustris	1600	96			Sanders 1969
	Gammarus fasciatus	2500	96			Sanders 1970
	Daphnia magna	100	48			"
	Cypridopsis vidua	320	48			"
	Asellus brevicaudus	2200	48			"
	Palaemonetes kadiakensis	2700	48			"
	Orconectes nais				100,000 μg/l 48 hr.	"
2,4-D (BEE)	CRUSTACEAN					
	Gammarus lacustris	440	96			Sanders 1969
	Gammarus fasciatus	5900	96			Sanders 1970
	Daphnia magna	5600	48			"
	Cypridopsis vidua	1800	48			"
	Asellus brevicaudus	3200	48			"
	Palaemonetes kadiakensis	1400	48			"
	Orconectes nais				100,000 μg/l 48 hr.	"
	INSECT					
	Pteronarcys californica	1600	96			Sanders and Cope 1968
2,4-D (BEE)	FISH					
	Pimephales promelas	5600	96	1500 μg/l lethal to eggs in 48 hour exposure	300 μg/l 10 mo.	Mount and Stephen 1967
2,4-D (IOE)	CRUSTACEAN					
	Gammarus lacustris	2400	96			Sanders 1969

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Table 29.
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Pesticide	Organism	Acute toxicity LC50 µg/liter	Acute toxicity LC50 hours	Sub-acute effects µg/liter	No effect µg/liter	Reference
2,4-D (DIETHYLAMINE SALT)	CRUSTACEAN					
	Gammarus lacustris					Sanders 1969
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970
	Daphnia magna	4000	48			"
	Crypidopsis vidua	8000	48			"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 µg/l 48 hr.	"
ENDOTHALL DISODIUM SALT	FISH					
	Pimephales notatus	110000	96			Walker 1964a
	Lepomis macrochirus	125000	96			"
	Micropterus salmoides	120000	96			"
	Notropis umbratilus	95000	96			"
	Micropterus salmoides	200000	96			Bond et al. 1960
	Oncorhynchus tschawytsch	136000	96			"
ENDOTHALL DIPOTASSIUM SALT	CRUSTACEAN					
	Gammarus lacustris				100,000 µg/l 96 hr.	Sanders 1969
	FISH					
	Pimephales promelas	320000	96			Surber and Pickering 1962
	Lepomis macrochirus	160000	96			"
EPTAM	CRUSTACEAN					
	Gammarus fasciatus	23000	96			Sanders 1970
FENAC (SODIUM SALT)	CRUSTACEAN					
	Gammarus lacustris	12000	96			Sanders 1969
	Gammarus fasciatus				100,000 µg/48 hr.	Sanders 1970
	Daphnia pulex	4500	48			Sanders and Cope 1966
	Simocephalus serrulatus	6600	48			"
	Daphnia magna				100,000 µg/l 48 hr.	Sanders 1970
	Cypridopsis vidua				100,000 µg/l 48 hr.	"
	Asellus brevicaudus				100,000 µg/l 48 hr	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais					
	INSECT					
	Pteronarcys californica	55000	96			Sanders and Cope 1968
	FISH					
	Lepomis	15000	48			Hughes and Davis 1962
HYAMINE 1622	FISH					
	Pimephales promelas	1600	96			Surber and Pickering 1962
	Lepomis macrochirus	1400	96			"
	Oncorhynchus kisutch	53000	96			Bond et al. 1960
HYAMINE 2389	FISH					
	Pimephales promelas	2400	96			Surber and Pickering 1962
	Lepomis macrochirus	1200	96			"
HYDROTHAL 47	CRUSTACEAN					
	Gammarus fasciatus	510	96			Sanders 1970
HYDROTHAL 191	CRUSTACEAN					
	Gammarus lacustris	500	96			Sanders 1969
	Gammarus fasciatus	480	96			Sanders 1970
HYDROTHAL PLUS	FISH					
	Lepomis macrochirus	3500	48			Hughes and Davis 1964
IPC	CRUSTACEAN					
	Gammarus lacustris	10000	96			Sanders 1969
	Gammarus fasciatus	19000	96			Sanders 1970
	Simocephalus serrulatus	10000	48			Sanders and Cope 1966
	Daphnia pulex	10000	48			"
KURON	CRUSTACEAN					
	Simocephalus serrulatus	2400	48			Sanders and Cope 1966
	Daphnia pulex	2000	48			"
MCPA	FISH					
	Lepomis macrochirus	1500	48			Hughes and Davis 1964
MOLINATE	CRUSTACEAN					
	Gammarus lacustris	4500	96			Sanders 1969
	Gammarus fasciatus	300	96			Sanders 1970
	Daphnia magna	600	48			"
	Asellus brevicaudus	400	48			"
	Palaemonetes kadiakensis	1000	48			"
	Orconectes nais	5600	48			"
MONURON	FISH					
	Oncorhynchus kisutch	110000	48			Bond et al. 1960
PARAQUAT	CRUSTACEAN					
	Gammarus lacustris	11000	96			Sanders 1969
	Simocephalus serrulatus	4000	48			Sanders and Cope 1966
	Daphnia pulex	3700	48			"
	INSECT					
	Pteronarcys californica				100,000 µg/l 96 hr	Sanders and Cope 1968
PEBULATE	CRUSTACEAN					
	Gammarus fasciatus	10000	96			Sanders 1970
PICLORAM	CRUSTACEAN					
	Gammarus lacustris	27000	96			Sanders 1969
	INSECT					
	Pteronarcys californica	48000	96			Sanders and Cope 1968

← WALKER 1964a

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Pteronarcys
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Table 29.
Continued.

Pesticide	Organism	Acute toxicity LC50 µg/liter	Acute toxicity LC50 hours	Sub-acute effects µg/liter	No effect µg/liter	Reference
PROPANIL	CRUSTACEAN					
	Gammarus fasciatus	16000	96			Sanders 1970
SILVEX (BEE)	CRUSTACEAN					
	Gammarus fasciatus	250	96			Sanders 1970
	Daphnia magna	2100	48			"
	Cypridopsis vidua	4900	48			"
	Asellus brevicaudus	40000	48			"
	Palamonetes kadiakensis	800	48			"
	Orconectes nias	60000	48			"
	FISH					
	Lepomis macrochirus	1100	48			Hughes and Davis 1963
SILVEX (PGBE)	CRUSTACEAN					
	Gammarus fasciatus	840	96			Sanders 1970
	Daphnia magna	180	48			"
	Cypridopsis vidua	200	48			"
	Asellus brevicaudus	500	48			"
	Palaemonetes kadiakensis	3200	48			"
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Lepomis macrochirus	16600	48			Hughes and Davis 1963
SILVEX (IOE)	FISH					
	Lepomis macrochirus	16000	48			Hughes and Davis 1963
SILVEX (POTASSIUM SALT)	FISH					
	Lepomis macrochirus	83000	48			Hughes and Davis 1963
SIMAZINE	CRUSTACEAN					
	Gammarus lacustris	13000	96			Sanders 1969
	Gammarus fasciatus				100,00 µg/l 48 hr.	Sanders 1970
	Daphnia magna	1000	48			
	Cypridopsis vidua	3200	48			
	Asellus brevicaudus				100,000 µg/l 48 hr.	Sanders 1970
	Palamonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 υg/l 48 hr.	"
	FISH					
	Oncorhynchus kisutch	6600	48			Bond et al. 1960
TRIFLURALIN	CRUSTACEAN					
	Gammarus lacustris	2200	96			Sanders 1969
	Gammarus fasciatus	1000	96			Sanders 1970
	Daphnia magna	560	48			"
	Daphnia pulex	240	48			Sanders and Cope 1966
	Simocephalus serrulatus	450	48			"
	Cypridopsis vidua	250	48			Sanders 1970
	Asellus brevicaudus	200	48			"
	Palaemonetes kadiakensis	1200	48			"
	Orconectes nais	50000	48			"
	INSECT					
	Pteronarcys californica	3000	96			Sanders and Cope 1968
VERNOLATE	CRUSTACEAN					
	Gammarus lacustris	1800	96			Sanders 1969
	Gammarus fasciatus	13000	96			Sanders 1970
	Daphnia magna	1100	48			"
	Cypridopsis vidua	240	48			"
	Asellus bravicaudus	5600	48			"
	Palaemonetes kadiakensis	1900	48			"
	Orconectes nais	24000	48			"

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Table 30.
Effects of various concentrations of herbicides on small bluegills and fry from four species of fish. From Hiltibran, 1967.

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Atrazine

Common name	Effect of herbicides on blue-gill[4]	ppm	Days survival of fry			
			Bluegill	Green sunfish	Lake chub-sucker	Small-mouth bass
Amitrole	25[1]	50[1]	Term[3]	Term	Term	—
Arsenite		15	7	Term	—	6
		8	Term	Term	—	Term
Atrazine (G)[2]	10	10	Term	Term	—	—
(WP)[2]	5	10	—	—	Term	3
Dalapon	50[1]	50[1]	Term	Term	Term	Term
Dichlobenil (G)	20	25	—	Term	Term	Term
		10	Term	—	—	—
Diquat cation	10	2.5	3	—	2	1
		1.3	—	—	2	4
Endothall (L)[2]	100	25[1]	Term	Term	Term	Term
(G)	50	10	Term	Term	Term	Term
Fenac	50	20	5	—	Term	—
Fenuron TCA (G)	20	10	Term	Term	Term	Term
Monuron TCA (G)	20	10	Term	5	Term	4
2,4-D		5	1	—	—	—
PGBE ester (G)	2	1	2	5	5	5
Mixed isopropyl-	3	4	2	Term	Term	—
butyl esters (L)		1	5	1	Term	1
2-ethylhexyl	50	10	Term	4	—	5
ester (G)		5	Term	—	Term	—
Dimethylamine salt (L)	40	25	Term	Term	Term	Term
Sodium salt	100					
2,4-DP						
Isooctyl ester (G)	20	10	Term	—	—	—
		1.5	—	—	Term	—
2,4,5-T						
Isooctyl ester (L)	1	4	—	1	—	0
		1	Term	Term	—	1
Isooctyl ester (G)	10	10	Term	Term	—	—
Sodium salt	50					
Silvex						
PGBE ester (L)	2	2.4	2	4	Term	—
		1.0	Term	—	Term	4
(G)	20	10	Term	Term	—	—
Potassium salt (L)	50	20	Term	—	Term	Term
(G)	30	10	—	Term	—	—
Sodium salt	50					
Simazine (G)	10	10	Term	7	Term	Term
(WP)	30	10	—	5	Term	3

[1] Highest rate used in these experiments, but below the reported toxic levels to fish.

[2] G = granular, L = liquid, WP = wettable powder.

[3] Terminated after 8 days of exposure with no loss of fry.

[4] Highest concentrations used which did not cause death of small bluegills during 12-day exposure.

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Diquat

Diquat is usually applied at a rate of 0.5 to 1.0 ppm over the area treated. It is applied as a spray to foliage of floating weeds or injected below the surface for control of submersed weeds. For most species this treatment should not have detrimental effects (Tables 29 and 32). Several authors, however, reported toxicity concentrations for certain organisms that approached these levels. Lawrence (1962) reported 0.5 + 0.5 mg/l as toxic to Pimephales promelas, Lepomis gibbosus, and Micropterus salmoides. His studies were accomplished under field conditions. The values he obtained for M. salmoides were not in agreement with the report of other authors (Tables 29 and 32). Hughes (1973) reported 1.0 mg/l was toxic to Morone saxatilis. Tatum and Blackburn (1965) reported values of 0.5 mg/l killed chironomids and oligochaetes. Wilson and Bond (1969) reported 0.048 mg/l was toxic to Hyalella azteca and Gilderhaus (1967) reported 1.0 mg/l was toxic to Daphnia pulex (Table 32). Hiltibran (1967) reported the effects of diquat cation on small bluegills and fry of other species (Table 30) and concluded that at recommended rates the chemical would not greatly effect survival of fish reproductive products.

Endothall

Sodium and potassium salts, aluminum oxides and acid of endothall are used for control of aquatic weeds. A derivative (dimethylamine salt) is especially effective for hydrilla control (Dumas 1976). This derivative is toxic to fish if it exceeds 0.2 mg/l in water. Dipotassium or disodium endothall is recommended for use at 5 mg/l and should be safe at this level (Dumas 1976). Hilitbran (1967) postulated that endothall would be one of the safest herbicides to use during the spawning season due to its low toxicity to fish (100 mg/l). Other authors have reported similar results (Tables 29 and 33).

Copper ($CuSO_4$)

Copper sulfate is used for algae control and can be toxic. Cairns (1974) lists toxicities of heavy metals including copper ($CuSO_4$) (Table 34) to 13 species of protozoa. Toxicities were quite low; however, the author does not stipulate the concentration in his tables. This author also presented dose-response curves for various protozoan species which indicated that species responded differently in a time sequence. He stated that differences in sensitivity might be explained if more were known about sensitivity of copper at the cellular level. Copper ion toxicities are also presented in Table 34. This ion is toxic to marine and freshwater organisms; however, toxicities often varied according to experimental conditions. For example, Palmer and Maloney (1955) reported that 0.18 ppm $CuSO_4$ in acute static bioassays was toxic (96 hr LC50) to Pimephales promelas. Tarzwell and Henderson (1960) found $CuSO_4$ toxicity varied according to water conditions: the compound was toxic at 0.05 ppm in soft water and 1.4 ppm in hard water. Doudoroff et al. (1966) used a copper cyanide complex in their tests and found this compound was toxic at 1.5 ppm. Mount (1968) reported copper to be toxic at 470 mg/l in continuous flow bioassays with P. promelas. Similar examples can been seen for other species, indicating the importance of standardized tests and data presentation.

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change 38.4 to 38.1

Table 31.

Residues of 2,4-D in fish from Loxahatchee National Wildlife Refuge. From Schultz and Whitney, 1974.

Station	Date	Species	Length, CM	Weight, G	Residues of 2,4-D, MG/KG[1]	Station	Date	Species	Length, CM	Weight, G	Residues of 2,4-D, MG/KG[1]
1	4-27-71	Redear sunfish	17.8	91	0.000	3	6-20-71	Gar	38.1	227	0.022
		(Lepomis	15.2	45					33.0	136	
		microlophus)							63.5	1090	
1	4-28-71	Largemouth bass	28.0	227	0.000				45.7	368	
		(Micropterus	28.0	228					43.2	368	
		salmoides)				1	6-20-71	Redear sunfish	20.3	—	0.000
1	4-28-71	Brown bullhead	30.5	454	0.000	3	6-24-71	Gar	38.1	590	0.000
		(Ictalurus	25.4	227					35.6	545	
		nebulosus)				3	6-24-71	Chubsuckers	27.9	368	0.000
1	4-28-71	Gar	35.6	182	0.000				30.5	409	0.000
		(Lepisosteus	33.0	136					25.4	318	0.000
		spp.)	30.5	91					25.4	318	0.000
			38.1	182					27.9	368	0.000
			48.3	454		1	7-1-71	Redear sunfish	21.5	227	0.022
3	4-28-71	Chubsuchers	33.0	368	0.000				17.8	113	
		(Erimyzon spp.)	27.9	272		2	7-2-71	Chubsuckers	33.0	499	0.000
			27.9	272					30.5	454	
3	4-28-71	Largemouth bass	27.9	227	0.000	2	7-2-71	Redear sunfish	20.3	182	0.000
			25.4	182					25.4	272	
3	4-28-71	Redear sunfish	20.3	227	0.000	2	7-2-71	Bluegill	22.9	182	0.000
			20.3	227				*(Lepomis*	20.3	182	
			17.8	136				*macrochirus)*	22.9	227	
1	6-17-71	Redear sunfish	17.8	91	0.101				20.3	182	
			17.8	91		3	7-2-71	Redear sunfish	20.3	182	0.010
			17.8	91					20.3	182	
			17.8	91		3	7-2-71	Chubsuckers	33.0	454	<0.010
1	6-17-71	Brown bullhead	27.9	343	0.012	1	7-15-71	Bluegill	20.3	136	0.000
			25.4	227					20.3	227	
			30.5	454					18.9	113	
			22.9	182		1	7-15-71	Redear sunfish	18.9	113	0.000
			22.9	182					22.9	294	
2	6-18-71	Chubsuckers	30.5	454	0.000	2	7-15-71	Chubsuckers	30.5	454	0.000
			25.4	227					33.0	567	
			25.4	227		2	7-15-71	Bluegill	17.8	227	<0.010
			27.9	272					17.8	204	
			22.9	91					20.3	227	
2	6-18-71	Gar	45.7	454	0.000				20.3	227	
			38.1	136					20.3	227	
			45.7	368		3	7-15-71	Bluegill	17.8	113	0.000
			33.0	91					20.3	227	
			35.6	182					22.9	340	
2	6-18-71	Redear sunfish	20.3	182	0.000				17.8	182	
			17.8	136					22.9	227	
			17.8	91		1	8-12-71	Gar	38.1	136	0.106
			22.9	272					48.3	454	
			17.8	91					43.2	272	
3	6-18-71	Chubsuckers	33.0	590	<0.010	1	8-12-71	Brown bullhead	38.1	681	<0.010
			33.0	454					30.5	318	
			30.5	409					27.9	272	
			27.9	227		1	8-12-71	Redear sunfish	17.8	91	<0.010
			27.9	227					17.8	91	
3	6-18-71	Redear sunfish	22.9	182	0.000				17.8	91	
			22.9	182					17.8	91	
			20.3	136					17.8	91	
			17.8	91		1	8-12-71	Chubsuckers	25.4	227	<0.010
3	6-18-71	Largemouth bass	35.6	454	0.000				27.9	272	
			27.9	318					27.9	318	
			27.9	318					25.4	227	
1	6-19-71	Gar	40.6	227	0.000				27.9	318	
			48.3	454		2	8-12-71	Redear sunfish	22.9	227	0.162
			55.9	454					22.9	227	
			33.0	136					17.8	91	
1	6-19-71	Redear sunfish	22.9	227	<0.010				22.9	227	
			20.3	136					17.8	91	
1	6-19-71	Brown bullhead	25.4	227	0.000	3	8-12-71	Chubsuckers	30.5	454	0.000
			25.4	227	0.000				33.0	567	
			27.9	318	0.000				25.4	227	
2	6-20-71	Redear sunfish	22.9	272	0.000				27.9	272	
			15.2	46					25.4	227	
			20.3	136		3	8-12-71	Largemouth bass	27.9	227	0.012

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Table 31.
Residues of 2,4-D in fish from Loxahatchee National Wildlife Refuge. From Schultz and Whitney, 1974.

Station	Date	Species	Length, CM	Weight, G	Residues of 2,4-D, MG/KG[1]
2	6-20-71	Gar	55.9	727	0.028
			40.6	227	
			48.3	368	
			38.1	227	
2	6-20-71	Brown bullhead	27.9	272	0.000
			25.4	227	
			27.9	227	
3	6-20-71	Largemouth bass	27.9	227	0.000
3	6-20-71	Chubsuckers	25.4	272	0.000
			27.9	272	
			25.4	272	
			22.9	182	
			25.4	227	
			25.4	227	
			27.9	272	
			27.9	227	
			27.9	368	
			30.5	368	
3	8-12-71	Redear sunfish	20.3	227	0.024
			20.3	227	
			22.9	368	
			17.8	136	
			17.8	136	
1	10-7-71	Brown bullhead	30.5	368	0.000
			34.3	545	
			35.6	681	
			38.1	681	
1	10-7-71	Gar	63.5	1362	0.000
			66.1	1644	
1	10-7-71	Redear sunfish	20.5	227	0.000
			20.3	182	
			20.3	227	
			19.1	182	
			24.2	368	
2	10-7-71	Redear sunfish	22.3	227	<0.010
			21.6	227	
			21.6	227	
			22.9	318	
			22.0	182	
2	10-7-71	Chubsuckers	36.9	726	0.000
			35.6	681	
			26.7	272	
			34.3	636	
			35.6	681	
3	10-7-71	Largemouth bass	35.6	636	0.000
			30.5	409	
			29.9	318	
			30.5	409	
			33.0	590	
3	10-7-71	Redear sunfish	22.9	227	0.070
			22.9	227	
			21.6	227	
			21.6	227	
3	10-7-71	Chubsuckers	33.0	545	0.000
			35.6	772	
			33.0	590	
			33.0	636	

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[1] When only one set of data is given for a series of fish, the fish were composited for analysis.

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Table 32.

Diquat: toxicity to aquatic organisms. From Folmar, 1977.

Organism	Type of test	Experimental conditions	Toxicity	Reference
Tendipedidae	L, ST, A	a	>100 T (96 h)	Wilson and Bond (1969)
Unnamed chironomids	FP	—	0.5 K (incomplete)	Tatum and Blackburn (1965)
Callibaetis spp.	L, ST, A	a	16.4 T (96 h)	Wilson and Bond (1969)
Lemnephilus spp.	L, ST, A	a	33.0 T (96 h)	Wilson and Bond (1969)
Daphnia pulex	L, ST, A	a,b,c,d,e	1.0 T (8 d) 3.0 K (8 d)	Gilderhaus (1967)
D. magna	L, ST, A	a,b,c,d,e,f	7.1 IC_{50} (26 h)	Crosby and Tucker (1966)
Hyalella azeteca	L, ST, A	a	0.048 T (96 h)	Wilson and Bond (1969)
Unnamed oligochaetes	FP	—	0.5 K (incomplete)	Tatum and Blackburn (1965)
Cardium edule	L, ST, A	—	>10.0 T (24 h)	Portmann and Wilson (1971)
Crassostrea virginica	L, CFT, A	—	1.0 NTE (96 h)	Butler (1965)
Enallagma spp.	L, ST, A	a	>100 T (96 h)	Wilson and Bond (1969)
Libellula spp.	L, ST, A	a	>100 T (96 h)	Wilson and Bond (1969)
Penaeus setiferus	L, CFT, A	—	1.0 NTE (48 h)	Butler (1965)
Crangon crangon	L, ST, A	—	>10.0 T (24 h)	Portman and Wilson (1971)
Aquatic insects, amphipods, copepods, ostracods	FP	—	1.0 NTE (7 d)	Hilsenhoff (1966)
Pimephales promelas	L, ST, A	a	10.0 NTE (96 h)	Lawrence et al. (1965)
P. promelas	L, ST, A	a,b,c,d,e	Soft water 14.0 T (96 h) Hard water 14.0 T (96 h)	Surber and Pickering (1962)
P. promelas	FP	—	0.5 + 0.5 Paraquat NTE	Lawrence (1962)
Fundulus similis	L, CFT, A	—	1.0 NTE (48 h)	Butler (1965)
Carassius auratus	FP	a,b,c,d,e	35.0 T (96 h)	Gilderhaus (1967)
Rasbora heteromorpha	L, ST, A	a,b,c,d,e	70.0 T (48 h)	Alabaster (1969)
Ictalurus punctatus (fry)	L, ST, A	a,b,c,d,e	10.0 NTE (72 h)	Jones (1965)
I. punctatus	L, ST, A	a	10.0 NTE (96 h)	Lawrence et al. (1965)
Lepomis cyanellus	FP	a,b,c,d,e	1,000 NTE	Yeo (1967)
L. gibbosus	FP	—	0.5 + 0.5 Paraquat NTE	Lawrence (1962)
L. macrochirus (fry)	L, ST, A	—	10.0 NTE (12 d)	Hiltibran (1967)
L. macrochirus (fingerlings)	L, ST, A	a,b,c,d,e	525 T (24 h) 150 T (48 h)	Hughes and Davis (1962)
L. macrochirus	FP	a,b,c,d,e	25.0 T (96 h)	Gilderhaus (1967)
L. macrochirus (fry)	L, ST, A	a,b,c,d,e	4.0 NTE (72 h)	Jones (1965)
L. macrochirus	L, ST, A	a	10.0 NTE (96 h)	Lawrence et al. (1965)
L. macrochirus	L, ST, A	a,b,c,d,e	Soft water 140 T (96 h) Hard water 140 T (96 h)	Surber and Pickering (1962)
Micropterus salmoides (fry)	L, ST, A	a,b,c,d,e	1.0 NTE (72 h)	Jones (1965)
M. salmoides	L, ST, A	a,b,c,d,e	Soft water 7.8 T (96 h)	Surber and Pickering (1962)
M. salmoides	FP	a,b,c,d,e	1,000 NTE	Yeo (1967)
M. salmoides	L, ST, A	—	11.0 T (48 h)	Muirhead-Thompson (1971)
M. salmoides	L, ST, A	a	10.0 NTE (96 h)	Lawrence et al. (1965)
M. salmoides	FP	a	0.5 + 0.5 Paraquat NTE	Lawrence (1962)

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Table 32.
Continued.

Organism	Type of test	Experimental conditions	Toxicity	Reference
Morone saxatilis (larvae)	L, ST, A	a,b,c,e	1.0 T (24, 48, 72, 96 h)	Hughes (1973)
M. saxatilis (fingerlings)	L, ST, A	a,b,c,e	35.0 T (24 h) 25.0 T (48 h) 15.0 T (72 h) 10.0 T (96 h)	Hughes (1973)
M. saxatilis (fry)	L, ST, A	a,b,c,d,e	35.0 T (24 h) 25.0 T (48 h) 15.0 T (72 h) 10.0 T (96 h)	Hughes (1969)
M. saxatilis (fingerlings)	L, ST, A	a	315 T (24 h) 155 T (48 h) 80.0 T (96 h)	Wellborn (1969)
Stizostedion vitreum	FP	a,b,c,d,e	2.1 T (96 h)	Gilderhaus (1967)
Esox lucius	FP	a,b,c,d,e	16.0 T (96 h)	Gilderhaus (1967)
Trout	—	—	20.0 T (24 h)	Holden (1964)
Onchorynchus tshawytscha	L, ST, A	—	29.0 T (48 h)	Muirhead-Thompson (1971)
Salmo gairdneri	L, ST, A	a	5.0 NTE (96 h)	Lawrence et al. (1965)
S. gairdneri	FP	a,b,c,d,e	11.2 T (96 h)	Gilderhaus (1967)
S. gairdneri	L, CFT, A	a,b,c,d,e	10.0 NTE; no avoidance	Folmar (1976)

1. Type of test, letters represent:
    - T = Toxicity test, used in conjunction with
        - ST = static
        - CFT = continuous flow
        - A = acute
        - C = chronic
    - L = Laboratory toxicity test
    - F = Field study, used in conjunction with
        - R = river, stream, or creek
        - M = marine
        - E = estuarine, and
        - O = other
2. Toxicity, active ingredient. All values in mg/1 (ppm) unless otherwise noted. Letters included with numerical values represent:
    - T = LC_{50} (also accompanied by a time factor, eg., 96 h)
    - K = Kill
    - SB = sublethal effects
    - NTE = No toxic effect
3. Experimental conditions. Factors reported in cited articles:
    - a = water temperature
    - b = pH
    - c = alkalinity
    - d = dissolved oxygen
    - e = dissolved solids
    - f = photoperiod

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Table 33.
Endothall: toxicity to aquatic organisms. From Folmar, 1977.

Organism	Type of test	Experimental conditions	Toxicity	Reference
Semotilus atromaculatus	L, ST, A	a,b,c,e	1,600 NTE (24 h)	Gillette et al. (1952)
Lepomis macrochirus	L, ST, A	a,b,c,e	428 T (24 h)[a] 268 T (48 h)	Davis and Hughes (1963)
Salmo gairdneri	L, CFT, A	a,b,c,d,e	10 NTE; no avoidance[a]	Folmar (1976)
Gammarus fasciatus	L, CFT, A	a,b,c,d,e	3.1 T (24 h)[b] 2.1 T (48 h) 0.48 T (96 h)	Sanders (1970)
Salmo gairdneri	L, ST, A	—	1.5 T (48 h)[b]	Cope (1965)
Lepomis macrochirus (fingerlings)	L, ST, A	a,b,c,d,e	0.8 T (24 h)[b] 0.8 T (48 h)	Hughes and Davis (1962)
L. macrochirus	RP	Pathological examination	S.B. 0.3 0.03 (28 d)[b]	Eller (1969)
L macrochirus (fry)	L, ST, A	a,b,c,d,e	0.75 NTE (72 h)[c]	Jones (1965)
L. macrochirus (fingerlings)	L, ST, A	a,b,c,d,e	0.3 T (24 h)[c] 0.3 T (48 h)	Hughes and Davis (1962)
L. macrochirus	L, A	—	376 (3-4 weeks) i.p. injection[c]	Walker (1964b)
Micropterus salmoides (fry)	L, ST, A	a,b,c,d,e	0.075 NTE (72 h)[c]	Jones (1965)
Ictalurus punctatus	L, ST, A	a,b,c,d,e	0.2 NTE (72 h)[c]	Jones (1965)
Daphnia magna	L, ST, A	a,b,c,d,e,f	46.0 IC_{50} (26 h)[d]	Crosby and Tucker (1966)
Gammarus lacustris	L, CFT, A	a,b,c,d,e	100 T (6 h)[d]	Sanders (1969)
G. lacustris	L, ST, A	a	>320 T (96 h)[d]	Nebecker and Gaufin (1964)
Mercenaria mercenaria (eggs)	L, ST, A	a	50.0 T (48 h)[d]	Davis and Hidu (1969)
M. mercenaria (larvae)	L, ST, A	a	>10.0 (10 d)[d]	Davis and Hidu (1969)
Crassostrea virginica	L, ST, A	a	>25.0 (12 d)[d]	Davis and Hidu (1969)
Notropis lutrensis	L, ST, A	a,b,c,e	95.0 T (96 h)[d]	Walker (1963, 1964a)
N. umbratilis	L, ST, A	a,b,c,e	105 T (96 h)[d]	Walker (1963, 1964a)
Pimephales notatus	L, ST, A	a,b,c,e	110-120 T (96 h)[d]	Walker (1963, 1964a)
P. promelas	L, ST, A	a,b,c,d,e	Soft Water 320 T (96 h)[d] Hard water 6.0 T (96 h)	Surber and Pickering (1962)
Erimyzon sucetta (fingerlings)	L, ST, A	—	25.0 NTE (12 d)[e]	Hiltibran (1967)

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Table 33.
Continued.

Organism	Type of test	Experimental conditions	Toxicity	Reference
Carassius auratus and *Cyprinus carpio*	L, ST, A	a,b,c,e	145-210 T (96 h)[e]	Walker (1963, 1964a)
Ictalurus punctatus (fry)	L, ST, A	a,b,c,d,e	100 NTE (72 h)[e]	Jones (1965)
I. nebulosus	L, ST, A	a,b,c,e	170-175 T (96 h)[d]	Walker (1963, 1964a)
I. melas	L, ST, A	a,b,c,e	180-185 T (96 h)[d]	Walker (1963, 1964a)
Lepomis cyanellus	FP	a,b,c,d,e	3.0 NTE[d]	Yeo (1970)
L. macrochirus (fry)	L, ST, A	a,b,c,d,e	50.0 NTE (72 h)[e]	Jones (1965)
L. macrochirus (fry)	L, ST, A	—	100 NTE (12 d)[e]	Hiltibran (1967)
L. macrochirus (fingerlings)	L, ST, A	a,b,c,d,e	Soft water 180 T (96 h)[d] Hard water 160 T (6 h)	Surber and Pickering (1962)
L. macrochirus (fingerlings)	L, ST, A	—	25.0 NTE (12 d)[e]	Hiltibran (1967)
L. macrochirus	L, ST, A	a,b,c,e	125-150 T (96 h)[d]	Walker (1963, 1964a)
L. macrochirus	L, ST, A	a,b,c,d,e	450 T (48 h)[e] 280 T (96 h)	Hughes and Davis (1965)
L. microlophus	L, ST, A	a,b,c,e	125 T (96 h)[d]	Walker (1963, 1964a)
Micropterus dolomieui (fingerlings)	L, ST, A	—	25.0 NTE (12 d)[e]	Hiltibran (1967)
M. dolomieui (fingerlings)	L, ST	—	10.0 NTE (12 d)[d]	Hiltibran (1967)
M. dolomieui	FP	a,b,c,d,e	3.0 NTE[d]	Yeo (1970)
M. salmoides (fry)	L, ST, A	a,b,c,d,e	10.0 NTE (72 h)[d]	Jones (1965)
M. salmoides (fingerlings)	L, ST, A	a,b,c,d,e	Soft water[d] 200 T (96 h)	Surber and Pickering (1962)
M. salmoides	L, ST, A	a,b,c,e	120-125 T (96 h)[d]	Walker (1963, 1964a)
Morone saxatilis (fingerlings)	L, ST, A	a,b,c,d,e	2,000 T (24 h)[d] 1,700 T (48 h) 710 T (96 h)	Wellborn (1971)
Notropis umbratilis	L, ST, A	—	40.0 NTE (21 d)[f]	Lindaberry (1961)
N. lutrensis	L, ST, A	—	40.0 NTE (21 d)[f]	Lindaberry (1961)
Pimephale notatus	L, ST, A	—	40.0 NTE (21 d) [f]	Lindaberry (1961)
Erimyzon sucetta (fingerlings)	L, ST, A	—	10.0 NTE (12 d)[f]	Hiltibran (1967)
Ictalurus punctatus	L, ST, A	a,b,c,d,e	50.0 NTE (72 h)[f]	Jones (1965)
Lepomis macrochirus	L, ST, A	—	100 NTE (21 d)[f]	Lindaberry (1961)
L. macrochirus (fry)	L, ST, A	a,b,c,d,e	2.0 NTE (72 h)[f]	Jones (1965)
L. macrochirus (fry)	L, ST, A	—	50.0 NTE (12 d)[f]	Hiltibran (1967)
L. macrochirus (fingerlings)	L, ST, A	—	10.0 NTE (12 d)[f]	Hiltibran (1967)
L. macrochirus	L, ST, A	a,b,c,d,e	650 T (48 h)[f] 280 T (96 h)	Hughes and Davis (1965)
L. cyanellus (fingerlings)	L, ST, A	—	10.0 NTE (12 d)[f]	Hiltibran (1967)
Micropterus salmoides	L, ST, A	—	10.0 NTE (21 d)[f]	Lindabrry (1961)
M. salmoides (fry)	L, ST, A	a,b,c,d,e	2.0 NTE (72 h)[f]	Jones (1965)
Salmon	L, ST	—	— NTE (21 d)[f]	Lindaberry (1961)
Salmo gairdneri	L, ST, A	—	10.0 NTE (21 d)[f]	Lindaberry (1961)

[a] di K salt formulation.
[b] TD-191 (mono-N, N dimethylmine salt) formulations.
[c] TD-47 (di-N, N dimethylmine salt) formulation.
[d] di Na salt formulation.
[e] di Na salt formulation — liquid.
[f] di Na salt formulation — granular.

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b dimethylmine
c dimethylmine

Table 34.

Lethal and tolerant concentrations of 12 toxic compounds tested on 13 species of Protozoa. From Cairns, 1974.

	Toxicant				
Test organism	Cr^{6+} (as $K_2Cr_2O_7$)	Phenol	Cu^{2+} (as $CuSo_4 \cdot 5H_2O$)	Pb_2 + [as $Pb(NO_3)_2$]	Mn^{7+} (as $KMnO_4$)
Chilomonas paramecium	1000 (>18)	1500 (560)	0.056 (0.024)	320 (5.6)	3.2; 5.6; 7.5 (1.0); (1.0); (1.0)
Peranema trichophorum	160 (100)	2500 (1000)	>100 (1.8)	(1000)	>32 (3.2)
Tetrahymena pyriformis	1000; >1000 (180); (750)	3200 (1000)	10 (0.32)	>100 (24)	3.2 (0.75)
Paramecium caudatum	5000 (1000)	10 (1.35)(<1.35)			
Paramecium multimicronucleatum	>1000 (320)	(1000)	0.1; >1.0 (0.032); (0.24)	56 (24)	10 (0.65)
Stentor coeruleus			1.0		
Euglena gracilis	>1000 (180)	>1000 (750)	>100; 500 (0.1); (5.6)	(1000)	10; 100 (3.2); (3.2)
Chlamydomonas sp.			56 (0.1)		
Chlamydomonas reinhardi			56 (18)		
Blepharisma	1000 (32)	(1000)	3.2; 3.2; 1.8 (0.1); (0.18); (0.32)	100 (42)	18; 5.6 (1.0); (0.32)
Amoeba proteus		(1000)	10 (1.0)		18 (<1.0)
Eglena acus			1.0 (0.18)		
Chaos carolinensis			2.4 (0.1)		

	Toxicant						
Test organism	Zn^{2+} (as $ZnSO_4 \cdot 7H_2O$)	Co^{2+} (as $CoCl_2 \cdot 6H_2O$)	Nitric acid	Acetic acid	Al^{3+} (as $AlCl_3 \cdot 6\ H_2O$)	Sn^{2+} (as $SnCl_2 \cdot 2H_2O$)	HCl
Chilomonas paramecium	>10; 3.2 (18); (5.6)	>2500 (1000)	10 (7.5)	100; 32 (180); (100)	2.4 (1.0)	10 (7.5)	
Peranema trichophorum	(1000)	>5000 (2500)	56 (5.6)	1000 (75)	>1000 (560)		
Tetrahymena pyriformis	5.6 (1.0)		18 (10)	320 (180)	3.2 (1.0)	32 (7.5)	3.2 (2.4)
Paramecium caudatum	32 (15.5)						
Paramecium multimicronucleatum	10 (0.56)		13.5 (10)				
Stentor coeruleus	42						
Euglena gracilis	1000 (5000)		(1000)	1000 (560)	(1000)	(100)	
Chlamydomonas sp.							
Chlamydomonas reinhardi	1.8 (1.0) (100)						
Blepharisma	100; 32; 56 (10); (5.6); (5.6)		32 (54)				
Amoeba proteus	(1000)						
Euglena acus							
Chaos carolinensis	>1000 (320)						

Although considerable data are available in the literature pertaining to toxicity and responses of organisms to toxic chemicals, conditions under which each test was run should be considered. Many factors can influence the responses of organisms to these materials. It is much easier to control environmental variables in the laboratory; it is impossible to control these variables in the field. When animals are subjected to bioassays, physical, and chemical factors of the environment as well as physiological mechanisms of the living organisms should be considered. Often the physiological responses, especially the cellular response of lower organisms, are not known. Physical and environmental variables including size, acclimation, water source, temperature, dissolved oxygen, and number of organisms per test container can be controlled in laboratory tests. Weiss and Botts (1957) studied effects of acclimation time, individual size, and temperature differences of three fish species, Pimephales promelas, Lepomis cyanellus, and Carassius auratus, when exposed to sarin (isopropyl methylposphonofluoridale). Generally, smaller fish consumed more oxygen than larger individuals. The size factor, however, was not strictly proportional among the groups in their response. Lepomis cyanellus was tested over a wider range (Figure 33). The T50 values increased by a factor of 1.6, whereas the oxygen consumption rate decreased by a factor of 1.57.

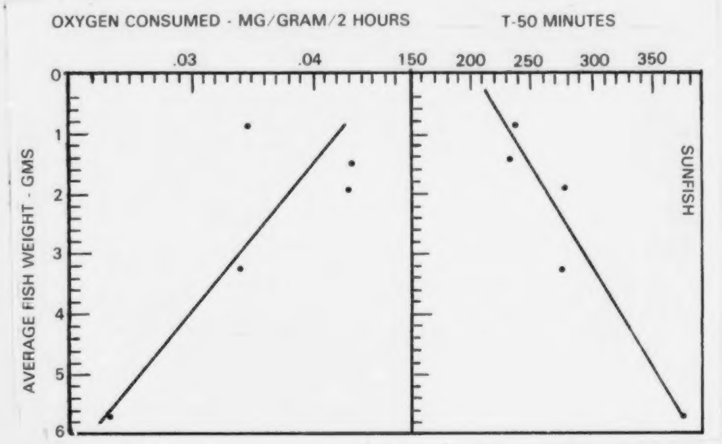

Figure 33. The effect of size of sunfish on time of response to sarin (10 ppb) and oxygen consumption. From Weiss and Botts, 1957.

The effect of time in laboratory stock tanks in response to sarin was determined (Figure 34). Over a period of three weeks, the T50 of all sizes increased, and oxygen consumption decreased (Weiss and Botts 1957). According to the authors, differences found in response to fishes at different times may have been due to some unknown seasonal factor. These authors also studied the influence of water temperature on oxygen consumption and concluded that temperature acclimatization prior to test exposure is necessary to allow for adjustment of the oxygen consumption rate.

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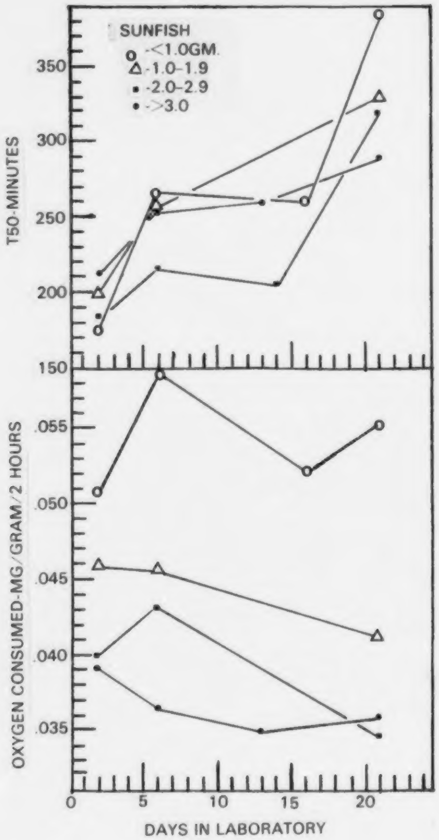

Figure 34. The effect of time in laboratory stock tanks on the time of response of sunfish to sarin (10 ppb) and oxygen consumption. From Weiss and Botts, 1957.

Dalapon

Dalapon ($C_3H_3CL_2O_2$) is used extensively for controlling ditchbank weeds and spot treatment of cattails. Toxicity of this compound to aquatic organisms is low, probable less toxic in most cases than 2,4-D. Toxicity values for dalapon for various aquatic animals are presented in Tables 29 and 35. These tables are literature values compiled by Anon. (1973) and Folmar (1977). Other authors report similar results. Chancellor and Ripper (1960) obtained an LC50 for brown trout (*Salmo trutta*) of 400 ppm. Kenaga (1973) tested 12 species of freshwater fish in static bioassays and found that LC50 was always greater than 100 ppm. Fish fry were resistant to dalapon (Table 35) and showed no mortality after exposure to 50 ppm. Johnson (1978) reported toxicity at 19 g/l for mosquitofish (*Gambusia affinis*).

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Table 35.
Dalapon: toxicity to aquatic organisms. From Folmar, 1977

Organism	Type of test	Experimental conditions	Toxicity	Reference
Simocephalus serrulatus	L, ST, A	a,b,c,e	16.0 IC_{50} (48 h)	Sanders and Cope (1966)
Daphnia pulex	L, ST, A	a,b,c,e	11.0 IC_{50} (48 h)	Sanders and Cope (1966)
Cardium edule	L, ST, A	—	>100 T (24 h)	Portmann and Wilson (1971)
Crangon crangon	L, ST, A	—	>100 T (24 h)	Portmann and Wilson (1971)
Pimephales promelas	L, ST, A	a,b,c,d,e	Soft water 390 T (96 h) Hard water 290 T (96 h)	Surber and Pickering (1962)
Erimyzon sucetta	L, ST, A	—	50.0 NTE	Hiltibran (1967)
Rasbora heteromorpha	L, ST, A	a,b,c,d,e	44.0 T (48 h)	Alabaster (1969)
Flounder	L, ST, A	—	>100 T (24 h)	Portmann and Wilson (1971)
Lepomis macrochirus (fry)	L, ST, A	—	50.0 NTE (12 d)	Hiltibran (1967)
L. macrochirus	L, ST, A	a,b,c,d,e	Soft water 440 T (96 h)	Surber and Pickering (1962)
L. macrochirus	L, ST, A	—	115 T (48 h)	Cope (1965)
L. cyanellus (fry)	L, ST, A	—	50.0 NTE	Hiltibran (1967)
Micropterus dolomieui (fry)	L, ST, A	—	50.0 NTE	Hiltibran (1967)
Trout	—	—	340 T (24 h)	Holden (1964)
Pteronarcys californica	L, CFT, A	a,b,c,d,e	100 NTE (96 h)[a]	Sanders and Cope (1968)
Crassostrea virginica	L, CFT, A	—	1.0 NTE (96 h)[a]	Butler (1965)
Penaeus aztecus	L, CFT, A	—	1.0 T (48 h)[a]	Butler (1965)
Fundulus similis	L, CFT, A	—	1.0 NTE (48 h)[a]	Butler (1965)
Cyprinus carpio	F, P	Pathological examination	SB 250, 25, 2.5 (28 d)[a]	Schultz (1971)
Rasbora heteromorpha	L, ST, A	a,b,c,d,e	240 T (48 h)[a]	Alabaster (1969)
Lepomis macrochirus (fingerlings)	L, ST, A	a,b,c,d,e	>1,000 T (24 h)[b] >1,000 T (48 h)	Hughes and Davis (1962)
Salmo gairdneri	L, CFT, A	a,b,c,d,e	1.0 SB Avoidance[a]	Folmar (1976)
Lepomis macrochirus	L, ST, A	a,b,c,d,e	>1,000 T (24 h)[c] >1,000 T (48 h)	Hughes and Davis (1962)

[a] Na salt formulation.
[b] Granular formulation.
[c] Wettable powder formulations.

A field experiment conducted by Brooker (1976) was designed to compare directly the ecological effects of the use of herbicides (dalapon and 2,4-D) and hand clearance for drainage channel maintenance of emergent weeds. He concluded that invertebrate densities generally remained similar during the two study years. No species were lost due to herbicide treatment. Some animals such as Herpetocypris, Calospectra, and Planorbia vorticulus increased in the herbicide treatment area. Replacement plant species (algae, submersed and floating-leafed species) apparently provided living space for these organisms. For additional information of the effects of dalapon on the environment, the review article by Kenaga (1973) should be consulted.

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Chronic Effects of Herbicides

The long-term or chronic effects of most herbicides have not been studied to any extent and pathophysiology of the chemicals has not been determined. Most toxicological studies have been conducted with laboratory mammals or live stock. Chronic or toxicological effects of herbicides on fish and wildlife are few.

Effects of 2,4-D on the eggs of the pheasant (Phasianus colchicus), red partridge (Perdix rufa), and gray partridge (Perdix perdix) were studied by Lutz-Ostertag and Lutz (1970). Eggs from these species were sprayed with 2,4-D at a concentration of 1 to 2 l/ha. They found the chemical to be toxic to embryos prior to the 19th day. In the majority of cases, surviving embryos suffered partial or total paralysis and exhibited lordosis, atrophied feet, and shriveled toes which remained clenched. Histological and cytological examination of the surviving embryos indicated sterility of half of the males and females.

Cope et al. (1970) studied chronic effects of 2,4-D on bluegill sunfish. Hematocrit values were highest at highest treatment concentrations but returned to normal after 3 days. Histopathological changes were noted primarily in the liver as parenchymal cells underwent shrinkage and loss of vacuolation. A decrease in glycogen was also noted. They reported that changes were most accentuated at the 14th day and were common at day 28 in the 5-10 ppm treatment lots. By day 56, cyological changes were still evident in the 10 ppm lot, but glycogen stores were nearly normal. Concomitant with the liver changes were also vascular abnormalities. Eosiniphilic deposits appeared throughout the vascular system. At least 30% of fish from the 0.1 ppm, 50% from the 0.5 ppm, 60% from the 1.0 ppm, and all fish from the 5 and 10 ppm treatments showed these symptoms at 72 hours. These deposits were present in 30% of the fish in the higher dosage at day 28, but were not found after 28 days. Circulatory stasis also occurred. After 84 days only occasional fish showed slightly increased capillary networks within the brain. Although a single exposure to 2,4-D in this study produced pathologic changes, all fish eventually returned to normal. As the authors point out, responses to repeated exposures and responses of more sensitive species are not known.

Eller (1968) studied effects of Hydrothol 191 on bluegills in Oklahoma ponds. He found morphometric and systematic changes in gills, liver, and testes. Gill filaments were fused on contact with the chemical at 0.3 ppm. After 14 days, gill structure returned to normal. Liver damage was reduced after 56 days and returned to normal by day 112. Hypertrophic cells appeared in the testes at the 3rd day and disappeared from fish from the 14th to 28th day. Blood dyscrasia occurred in fish from ponds treated with 0.5 ppm and existed for 36 days.

These studies indicate that fish suffer pathological changes when exposed to herbicides. These changes may be severe enough to cause death or physiological changes that last for a period of time and then recover. A question can be posed as to whether fish exposed to subacute levels continuously would eventually die. That possibility exists.

Chemicals added to the water might also influence fish behavior. Folmar (1976) exposed rainbow trout (Salmo gairdneri) to concentrations of various herbicides. Trout detected sub-lethal concentrations and avoided the treatment area. The lowest concentrations of each chemical avoided were 0.0001 mg/l $CuSO_4$, 0.1 mg/l xylene and acrolein, 1.0 mg/l dalapon and 2,4-D, and 10.1 mg/l glyphosate, dipotassium salt of endothall, and diquat. In most cases, fish detected these herbicides at below toxic concentrations. The author postulated that although the chemicals were not toxic to fish, they could have influenced habitat selection.

Another factor that should be considered as a chronic or long-term effect is the influence chemical control could have on food chains. From the previous discussion pertaining to acute toxicity and LD50, certain zooplankton have tolerance levels which could make them susceptible to herbicide treatments. Reduction or eradication of these organisms at a time when larva fishes are dependent upon them for food could seriously affect a year class and cause a shift in fish species. It seems logical, therefore, to determine food webs and to determine the reliance of fishes on species that could be intolerant to herbicidal chemicals.

Plant Destruction

Data presented in this paper indicate that aquatic herbicides are relatively non-toxic to aquatic animals and their residues in water and soils are short-lived. Water quality parameters are usually not affected and if differences are seen, they are not long-lasting. The most detrimental effect could be establishment of algal blooms. These blooms, the result of nutrient release by decaying weeds, are an indirect effect of the treatment. The degree of severity depends upon the nutrients within the system and the amount of weeds killed. Literature data indicate that algal blooms do not create problems in large systems or systems through which water flows. Problems occur in small systems such as aquaria, swimming pools, and small ponds where there is very little water circulation and redeposition of nutrients in the soil. Carter and Hestand (1977) investigated effects of various herbicide formulations in plastic pools on phytoplankton succession. They concluded that liquid herbicides caused a fast kill of plants and release of nutrients, resulting in an explosive growth of phytoplankton and bacteria (Table 36). System E (a pelleted form of endothall) was the least detrimental of the herbicides. Due to its slow release, phytoplankton growth was gradual. In all cases, except System E, phytoplankton and bacteria populations returned to normal at approximately 84 days after treatment.

Further evidence for the nonimpact of herbicides on aquatic animals is seen in a pond study by Simpson and Pimental (1972). The authors used 12 0.1 ha ponds treated and untreated with Fenac. Short-term effects were depression of dissolved oxygen concentration, an increase in community respiration, and reduction of carbonate ion one month after treatment. These effects were short-lived and did not seem to affect phytoplankton and zooplankton populations. The authors stated that this particular chemical did not deter algal growth, particularly Chara, which may have buffered the system from drastic change.

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Table 36.
Toxicity of copper to marine and aquatic life. From Anon., 1973.

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature Citation*
Copper Cu	0.57 mg/l (2 hr)	Wattersipora	copper sodium citrate pH 7.0-8.2	Wisely and Blick 1967
	3.65 mg/l (2 hr)	Bugula	copper sodium citrate pH 7.0-8.2	Wisely and Blick 1967
	0.51 mg/l (2 hr)	Spirorbis	copper sodium citrate pH 7.0-8.2	"
	2.9 mg/l (2 hr)	Galeolaria	copper sodium citrate pH 7.0-8.2	"
	22.5 mg/l (2 hr)	Mytilus	copper sodium citrate pH 7.2-8.2	"
	0.4-0.5 ppm (2 day)	Salmo gairdneri	static acute bioassay; a,c,d,e,f	Brown 1968
	1.25 ppm	Lepomis macrochirus	static acute bioassy, a,c,d,e; Cu++; fish acclimatized 2 wks. in syn. dil. water.	Cairns Jr. and Scheier 1968
	1.04 ppm	"	static ;acute bioassay; a,c,d,e; fish acclimatized 2 wks in syn. dil. water copper-	"
	26.0 ppm	Lepomis macrochirus	copper acelic acid; all fish acclimatized 2 wks. in syn. dil water.	Cairns Jr. and Scheier 196P
	5.2 ppm	"	a,c,d,e; static acute bioassay same as above except that copper-acetaldehyde was used.	"
	5.2 ppm	"	same as above except that acetone; copper mixture was used	"
	430 mg/l	adult minnows	static test	Mount 1968
	470 mg/l	Pimephales	continuous flow bioassay	Mount 1968
	1.0 ppm (6.5 day)	Gasterosteus aculeatus	static acute bioassay; a,c; using $Cu(NO_3)_2$	Jones 1938
	0.23 mg l (6 hr)	Balanus balanoides	hypertonic seawater	Pyefinch and Mott 1948
	0.46 mg/l (6 hr)	" crenatus	hypertonic seawater	"
	3.3 mg/l (24 hr)	Orizias	$CuCl_2 2H_2O$	Doudoroff and Katz 1953
	0.74 ppm	Lepomis macrochirus	static acute bioassay; a,c,d,e, distilled aerated water	Trama 1954
	7.0 mg/l (48 hr)	"	20 C; pH 8.3	Turnbull et al. 1954
	0.18 ppm	Pimephales promelas	static acute bioassay; a,c,d,e,f; $CuSO_4$	Palmer and Maloney 1955
	84 ppm (2 day)	Gambusia affinis	static acute bioassay; turbid water; a,c,d,e,g; $CuSO_4$	Wallen et al. 1957
	75 mg/l	Gambusia affinis	24-27 C; using copper sulfate in highly turbid water	Wallen et al. 1957
	56.000 ppm (2 day)	Gambusia affinis	cupric oxide; static acute bioassay a,c,d,e,g; turbid water 19-20 C	Wallen et al. 1957
	38 ppm (1 day)	Salmo gairdneri (fry)	$CuSO_4$; a,c,e,f,i,p; static acute bioassay	Turnbull-Kemp 1958
	1.25 mg/l (time not given)	Lepomis macrochirus	in soft water; 18-20 C; $CuCl_2$	Academy of Natural Sciences 1960
	48 hr.	Daphnia magna		Cabejszek and Stasiak 1960
	1.9 ppm	Japanese oyster	Copper sulfate	Fujiya 1960
	1.9 mg/l	oysters	pH 8.2; 12 C	Fujiya 1960 1961
	1.4 ppm	Pimephales promelas	static acute bioassay; a,c,d,f, hard water; $CuSO_4$	Tarzwell and Henderson 1960
	0.05 ppm	"	same as above using soft water	"
	10 ppm	Lepomis marcrochirus	same as above using hard water	"
	0.2 ppm	"	same as above using soft water	"
	1.9 mg/l	oysters	$CuCl_2 \cdot 2H_2O$	Fujiya 1961
Copper (Cu)	0.40 ppm	Limnodrilus hoffmeisteri	static acute bioassay; hard water $CuSo_4$ a,c,d,i	Wurtz and Bridges 1961
	0.425 ppm	Gyraulus circumstriatus	static acute bioassay; a,c,d,i; hard water; $CuSO_4$	"
	0.27 ppm	Physa heterostropha	same as above	"
	1.5 mg/l (2-3 d)	Nereis		Raymount and Shields 1962
	0.27 ppm	Physa heterostropha	21 C hard water as $CuSO_4$	Wurtz 1962
	0.050 ppm	"	same as above; young	"
	0.56 ppm (1 day)	"	static acute bioassy; a,c,f; $CuSO_4$ hard and soft water.	Wurtz 1962
	90 ppm (1 day)	Carassius auratus Poecilia reticulata	conc. as copper sulfite	Floch et al. 1963
	15 ppm (1 day)	toad and frog tadpoles	Conc. as copper sulfite	Floch et al. 1963
	10 ppm (2 days)	"	"	"
	5 ppm (3 days)	"	"	"
	20 ppm (3 day)	Dragon fly larvae	conc. as copper sulfite	Floch et al. 1963
	40 ppm (1 day)	"	"	"
	2 ppm (1 day)	daphnia longispina	"	"
	0.1 ppm	Nereis virens	time not specified	Raymount and Shields 1963
	2 ppm (2 hr)	Salmo gairdneri	$CuSO_4 \cdot 5H_2O$	Herbert and Van Dyke 1964
	0.2 ppm (48 hr)	"	"	"
	1.5 ppm	Gammarus lacustris	static acute bioassay; a,e, $CuSo_4$	Nebeker and Gaufin 1964
	19 ppm (12 days)	Nereis virens	time not specified	Raymount and Shields 1963
	0.980 ppm	Lepomis macrochirus	$CuCl_2$	Cope 1965
	2.8 ppm	Lepomis macrochirus	static acute bioassay; $CuSO_4$a	"
	0.8 ppm (2 day)	Salmo gairdneri	a,c,e,f,l,m; field study in a river	Herbert et al. 1965
	0.034 ppm (1 day)	Salmo salar	continuous flow, acute bioassay g,c,f; with 3 μg l Zn and 2 μg/l Cu	Sprague 1965
	32 μg/l (time not given)	juvenile salmon	in very soft water (14 mg l hardness)	Sprague and Ramsey 1965
	0.150 ppm (2 day)	Salmo gairdneri	static acute bioassay, a, $CuSO_4$	Cope 1966
	2.800 ppm (2 day)	Lepomis macrochirus	same as above	Cope 1966
	1.5 ppm	Pimephales promelas	as CN using copper cyanide complex; static acute bioassay; a,c,; soft water	Doudoroff et al. 1966
	1.2 ppm	"	same as above except conc. as Cu	"
	1.14 mg/l	Pimelometopon pulchrum	in hard water; $CuSO_4 \cdot 5H_2O$	Pickering and Henderson 1966
	10.2 mg/l	Lepomis macrochirus	in hard water "	"
	0.048 ppm	Salmo salar	BSA; a; incipient lethal level with 0.600 Zn	Sigler et al. 1966
	3.0 ppm	Orconectes rusticus	continuous flow acute bioassay, a,c,e,f; 20 C; intermolting stage	Hubschman 1967
	1.0 ppm (1 day)	"	same as above; adult crayfish used	"
	1.0 ppm (6 day)	"	same as above; juvenile crayfish used	"
	1.0 ppm (6 day)	"	same as above; recently hatched young which remained clinging to pleopods of female during 1st molt were used.	"
	0.25 mg/l	Oroconectes rusticus embryo	time not given	Hubschman 1967

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Table 36.
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Constitutent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
	84.0 μg/l	Pimelometopon pulchrum	soft water; static bioassay	Mount and Stephen 1969
	75 μg/l	"	" continuous flow bioassay	"
	0.795-0.815 ppm (5 day)	Nitzschia linearis	static acute bioassay, a,c,e; $CuCl_2$	Patrick et al. 1958
	1.25 ppm	Lepomis macrochirus	same as above	"
	0.2 mg/l (48 hr)	Penaeus duorarum	in the dark; 15 C; $CuSO_4$	Portmann 1968
	30 mg/l (48 hr)	Penaeus aztecus	"	"
	100 mg/l "	shore crab	"	"
	1 mg/l "	cockle	"	"
	430 μg/l	Pimelometopon pulchrum	static bioassay; hard water	Mount and Stephen 1969
	470 μg/l	"	continuous flow bioassay; hard water	Mount and Stephen 1969
	1.7 mg/l	Capeloma decisum	soft water	Arthur and Leonard 1970
	0.039 mg/l	Physa integra	soft water	Arthur and Leonard 1970
	0.20 mg/l	Gammarus pseudolimnaeus	soft water	"
	48 hr	Salmo gairdneri		Brown and Dalton 1970
	3.2 mg/l	Fundulus heteroclitus	20-22 C; no feeding during the 96 hrs. aerated water	Jackim et al. 1970

Type should follow typography on p. 99

L = Laboratory bioassay
  BS = bioassay static
  BCF = bioassay continuous flow
  BA = bioassay acute
  BCH = bioassay chronic
a = water temperature
b = ambient air temperature
c = pH
d = alkalinity (total, phenolphthalein or caustic)
e = dissolved oxygen
f = hardness (total, carbonate, Mg or CaO)
g = turbidity
h = oxidation reduction potential
i = chloride as Cl
j = BOD, 5 day; (J) = BOD, short-term
k = COD
l = Nitrogen (as NO_2 or NO_3)
m = ammonia nitrogen as NH_3
n = phosphate (total, ortho-, or poly)
o = solids (total, fixed, volatile, or suspended)
p = CO_2
  BOD = biochemical oxygen demand

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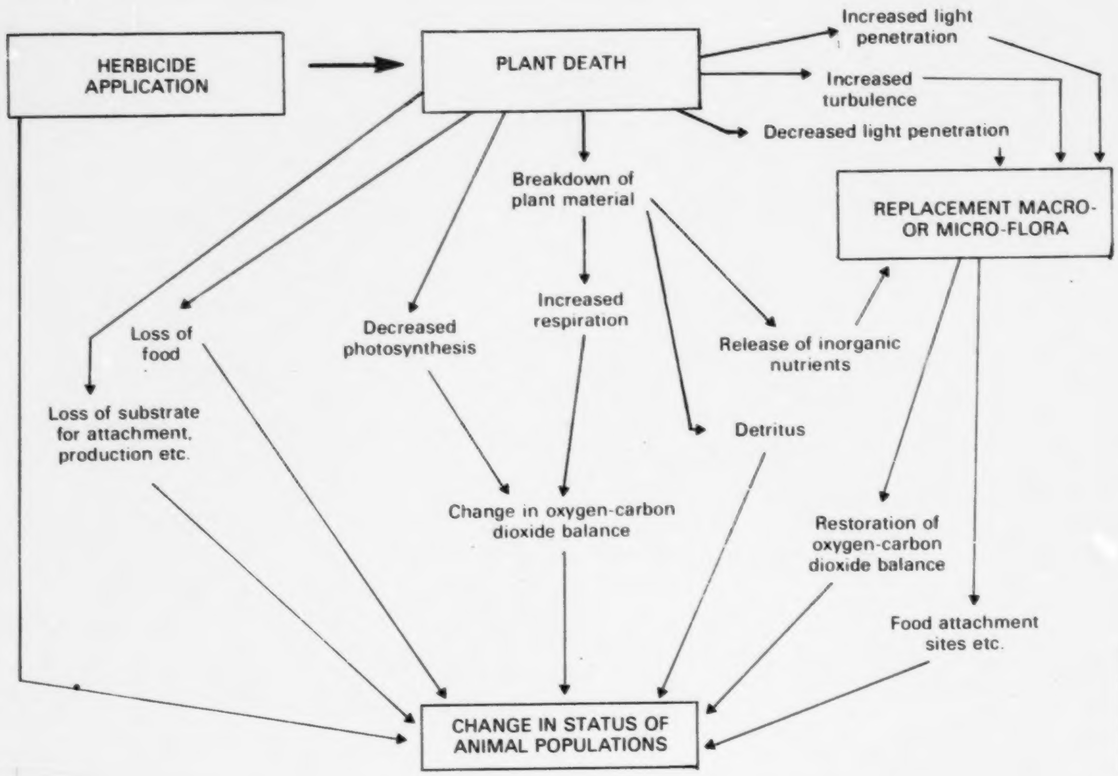
HERBICIDE APPLICATION
PLANT DEATH
Increased light penetration
Increased turbulence
Decreased light penetration
REPLACEMENT MACRO- OR MICRO-FLORA
Breakdown of plant material
Loss of food
Decreased photosynthesis
Increased respiration
Release of inorganic nutrients
Loss of substrate for attachment, production etc.
Detritus
Change in oxygen-carbon dioxide balance
Restoration of oxygen-carbon dioxide balance
Food attachment sites etc.
CHANGE IN STATUS OF ANIMAL POPULATIONS

Figure 35. Effects of herbicide application and the destruction of submerged plants likely to be of consequence in determining faunal changes. From Brooker and Edwards, 1975.

Destruction of aquatic vegetation can lead to continued phytoplankton blooms as evidenced by Lake Apopka, Florida. This large lake (12,500 ha, average depth approximately 2 m) was formerly a clear-water lake with luxuriant vegetation and was noted for excellent bass fishing. At the present time, the lake experiences continued algal blooms (EPA 1979). Several causes are given for this change: 1) nutrient overloading; 2) vegetation removal in 1947 by a hurricane; 3) hyacinth spraying and fish poisonings which added nutrients; and 4) water level stabilization. The first algal bloom was noted after vegetation was removed in 1947. The buffering capacity (nutrient removal and tie-up) was removed when the plants no longer existed. It is interesting to note that one of the proposed beneficial effects after lake drawdown will be the establishment of macrophyte growth. These plants will compete with phytoplankton for nutrients and thus reduce algal blooms (EPA 1979).

Brooker and Edwards (1975), in a review paper, discussed changes that can occur after aquatic weeds are killed (Figure 35). Most changes are due to decay and disappearance of the weeds. These changes can be "good" or "bad," depending upon the user group. The user group, however, often changes their attitude once a change is made and becomes permanent.

Fish populations can be affected by removal of macrophytes. Many fish depend upon organisms that are associated with vegetation, while others are dependent upon vegetation for spawning sites. Data, however, pertaining to changes in fish productivity are conflicting when a water body is treated with herbicides. Bennett (1971) reported data that indicated productivity may increase or decrease, depending on the situation.

Fishery biologists, especially in Florida, agree that vegetation is needed to maintain good fish populations, but the optimal amount of vegetation needed is not known. Data from Orange Lake, Florida, indicate that when 80% of the lake was covered with hydrilla, many small fish abounded in the vegetation. When the vegetation decreased the next year, fewer small fish were present, but growth and population structures improved (Shireman and Haller 1979). Total vegetation infestation of the water column probably exerts a physical constraint on foraging efficiency by eliminating the gradient or ecotone between open water and submersed macrophytes. Colle and Shireman (1980) found that bluegill and redear sunfish condition factors and weight-length relationships were not adversely affected until hydrilla occupied the majority of the water column. Elimination of the forage gradient was given as a possible cause. Harvestable largemouth bass exhibited low condition values once hydrilla coverage was above 30%; however, smaller bass were not adversely affected until coverage exceeded 50%. Shireman et al. (1979) compared condition factor values for bluegill and redear sunfish from Lake Wales during two years of heavy hydrilla infestation with a central 50% range of K(TL)'s from Carlander (1977). Condition for both bluegills and redears only approached or exceeded Carlander's mean value during 1977 when hydrilla coverage was reduced.

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Mechanical Harvesting

Removal of excessive aquatic plant growth from the aquatic environment often enhances the usefulness of the water. Mechanical harvesting is often considered the most environmentally safe method of removing aquatic plants, particularly aquatic macrophytes, and has been considered as an important method for removing nutrients (McNabb and Tierney 1972). Burton et al. (1979) upon reviewing data from a range of lakes suggest that aquatic plant harvesting has very little effect on reducing the nutrient content of many lakes. In fact, they noted that recent research has shown that many rooted aquatic macrophytes extract nutrients from the sediments (McRoy and Barsdate 1970; Bristow and Whitcomb 1971; DeMarte and Hartman 1974; Bole and Allan 1978) and that cutting or damaging the stems allows the macrophytes to pump nutrients to the overlying water (DeMarte and Hartman 1974; Carpenter and Adams 1977; Carpenter and Gasith 1978; Bole and Allan 1978), thus increasing plant nutrient concentrations. This nutrient release often leads to development of phytoplankton blooms or to growth of filamentous algae (Nichols 1973; Burton et al. 1979). Decay of plant material not removed from the water during harvesting can also release nutrients to the water column (Jewell 1971; Nichols 1973) and can further support algal growth.

Harvesting macrophytes can also affect the aquatic environment in other ways. Removal of rooted macrophytes destablizes bottom sediments, thus allowing resuspension of bottom sediments. This not only increases turbidity, but increases the rate of erosion in the littoral zone. Resuspension of organic materials can also decrease oxygen concentration in the water column (Jewell 1971). Harvesting may also result in diminished growth of macrophytes in subsequent years (Neel et al. 1973; Nichols 1974) by decreasing rate of input of detrital organic matter which is an important food source for many fishes and benthic organisms. Reduced plant growth will also result in a loss of substrate for attached algae and cover for invertebrates and fish (Burton et al. 1979).

Biological Control

Management of aquatic plant growth, through use of living organisms, offers a useful approach to managing the consequences of nutrient enrichment. Shapiro (1979) suggested that phytoplankton populations could be reduced by manipulations of fish populations. Shapiro cited studies that indicated that loss of fish populations through either winter kill (Schindler and Comita 1972) or chemical removal (Bandow 1978) often resulted in significant reduction in algal biomass with a corresponding increase in water transparency. Decline in algal biomass nearly always corresponds to large increases in *Daphnia* populations. Large *Daphnia* seem to remove algae through grazing (Burns 1969). However, Bandow (1978) noted that decline in algal abundance could be attributed to decrease in bottom-feeding fish that pump nutrients to the overlying water during feeding (Lamarra 1975a, 1975b).

Shapiro (1979) suggested that complete removal of fish populations was probably not necessary to obtain reduction in algal populations. Brooks and Dodson (1965) showed that zooplanktivorous fish select larger zooplankton during feeding. Shapiro (1979) suggested selective reduction in population size of zooplanktivorous fish through use of chemicals or the stocking of large predators that could reduce predation on the larger forms of zooplankton. This would permit these zooplankton to exert greater grazing pressure on algae.

Fish can also exert a degree of control on aquatic macrophytes. Rose and Moen (1953) reported that large populations of carp (*Cyprinus carpio*), buffalo (*Ictiobus spp.*), and sheepshead (*Aplodinotus grunniens*) suppressed growth of macrophytes in Lake East Okoboji, Iowa. They noted that after removal of 2,726 lb rough fish/acre, biomass of bottom vegetation increased. The feeding activities of the fish apparently uprooted vegetation. Recent studies by Lamarra (1975a, 1975b) suggest that carp pump nutrients which can support algal growth into the waters. Since Lake East Okoboji had large numbers of blue-green algae, shading by algae might also have contributed to reduction in plant growth. In fact, Smith and Swingle (1941a, 1941b) recommended fertilization of waters to reduce aquatic macrophyte growth.

Perhaps one of the most promising fish for management of aquatic macrophytes is the grass carp (*Ctenopharngyodon idella*). Possible introduction of this fish, however, has caused considerable concern among many biologists. A main concern is that the introduction of non-native fish will reduce population size of native fish. Although Rose and Moen (1953) noted that common carp caused reduction in game fish, grass carp normally do not spawn in lakes, and thus population size can be controlled through stocking. These fish, however, eat large amounts of vegetation and 100% removal of vegetation could have important indirect effects on the aquatic environment. Grass carp can release plant nutrients through feeding (Hickling 1966; Michewicz et al. 1972) and it is possible that very rapid removal of large amounts of macrophytes by these fish could result in algal blooms. If fish populations are controlled, however, algal blooms probably will not occur. If algal blooms occur, they might reduce populations of sight-feeding fish. Most studies, however, indicate there is no direct competition between grass carp and other fish (Kilgen and Smitherman 1971; Rottmann 1976).

The effect of grass carp on water quality has not been fully investigated. Studies by Lembi et al. (1978) in ponds indicate certain water quality changes. Dissolved potassium concentration increased in all ponds stocked with the fish and appears to be the water quality parameter most sensitive to vegetation alteration. Nitrogen and phosphorus concentrations were not statistically different after stocking. Shireman et al. (1979) found similar results in a 300-acre Florida lake stocked with grass carp. Potassium concentration increased three to four times when vegetation biomass was reduced. Studies (Michewicz et al. 1972) on small enclosures stocked with grass carp have shown an increase in water hardness and nitrogen but not in phosphorus. Studies by Haller and Sutton (1976), however, reported an increase in phosphorus during rapid hydrilla control by grass carp. They did not, however, detect changes in water turbidity. Other authors (Terell 1974;

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Forrester and Lawrence 1978) reported no changes in water quality after grass carp were stocked. Terrell (1974) found significant increase in sediment concentrations of iron, magnesium, and orthophosphate; calcium and magnese did not change.

Research Needs for Development of Effective, Environmentally Safe Aquatic Plant Management Programs.

Private citizens, water resource managers, politicians, some aquatic research scientists, and others, after examining the literature review in this paper, might argue that there is no need for additional research as an overabundant accumulation of facts already exists. They will agree that all monies should be expended on aquatic plant control. We agree that there is an embarassingly large accumulation of data and that, if sufficient money is available, we have the knowledge and techniques to kill aquatic plants. Most aquatic weeds, however, are controlled by chemicals that environmental groups and the public at large find unacceptable. It is necessary to provide alternative control methodology and to document any potential hazard in using certain chemicals in water. An outright ban on chemical control of weeds would create health, recreational, economic, and flood problems. Development of environmentally safe, effective aquatic plant management programs will require a better understanding of how aquatic systems function. The major research need, therefore, is the advancement of our knowledge concerning functioning of aquatic systems influenced by aquatic management techniques.

If, with the accumulated data now available, we still need additional information before we can develop effective, yet safe aquatic plant management programs, the problems will involve how to advance our knowledge. Certainly, a poll of research scientists would show that there are numerous research questions, and, hence, reseach projects that should be funded.

We feel that the following questions should be considered and researched:

Plants and Water Quality

1. Are current river and lake classification systems appropriate? Should separate classifications be established for specific user groups? Should we attempt to provide multiple-use waters? Should separate classifications be established for aquatic plant management? Water bodies, in most cases, have a primary use: for example, canals for water transport, reservoirs for potable water, fish management lakes, and suburban lakes for general recreational uses. These water bodies, because of their various functions, probably need different management programs that may be mutually exclusive.

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2. What are the standards against which water quality and aquatic weed nuisances should be judged? User groups often establish their own standards without regard for biological or ecological principles.

3. Are water quality measurements important as impact measurements? How does water quality change with changes in the natural physical, chemical, and biological characteristics of aquatic systems? Methods must be developed for evaluation of natural water quality changes versus changes caused by management programs.

4. What are the relationships between plant nutrient concentrations in lakes and subsequent aquatic macrophyte biomass?

5. Will the degree of an algal bloom and its persistence depend on the lake trophic state? Can we expect all lakes to react in a similar manner regardless of trophic state?

6. Are there critical algal and aquatic macrophyte concentrations that can be achieved to minimize problems in a given lake? Can a balance be reached where neither of these plants causes problems?

7. What physical, chemical, and biological factors control total aquatic plant growth and growth of specific species of aquatic plants? Can natural and/or anthropogenic factors, such as nutrient inputs, be manipulated to enhance growth of desirable species and discourage growth of non-beneficial aquatic plants?

Aquatic Organisms

8. What is the effect of aquatic plant biomass on water quality, zooplankton, benthos, and fish?

9. At what biomass do aquatic plants, such as hydrilla or blue-green algae, become detrimental to fish populations?

10. Can we manage problem aquatic macrophytes, such as hydrilla, or beneficial plants, such as panicum, to satisfy maximum fish and wildlife needs?

11. Are there certain species of aquatic plants, such as panicum or green algae, that can be managed to enhance fish populations.?

12. Can research develop genetically superior and hardier aquatic plants? Emergent or floating plants in certain situations might be more beneficial than submersed plants. Genetically superior plants might compete more effectively for nutrients, causing a reduction in less desirable plants. Plants could be selected to enhance fish production.

Biological Control

13. Should there be a moratorium on introduction of potential biological control organisms until better methods are developed and the potential behavior of these organisms in U.S. ecosystems can be predicted? More exhaustive searches should be made to determine if biological control organisms are present in U.S. waters. Evaluation should be made to determine effectiveness of existing biological controls for different management programs. It is our feeling that many of the current management strategies have not been developed for their use.

14. Can grass carp be effectively managed alone to control vegetation? Data indicate that considerable time and effort must be expended to assure that desired control is achieved. Grass carp stocked alone often do not achieve desired results.

15. Can we begin ichthyofauna reconstruction in our nation's waters to enhance aquatic plant management programs and thus control aquatic problems with minimum energy inputs and environmental degradation? Herbivores and other fishes that feed on algae or other organisms could be utilized to reconstruct entire systems.

Chemical Control

16. Considering there is a spectrum of toxicity and persistence, is it best to use a persistent chemical once every few years or a non-persistent herbicide 3 or 4 times yearly? The persistent herbicide would reduce the volume of weeds killed each year and would probably decrease environmental problems such as those associated with weed decay.

17. What are the short-term and long-term impacts of chemical usage in aquatic systems? Are there long-term chronic effects on non-target organisms that are not detected by current bioassay techniques? Is the continued use of non-persistent, non-toxic herbicides detrimental to fish and other aquatic organisms?

18. What is the best time to apply herbicides to control aquatic weeds and reduce impacts on non-target organisms? Can treatments be timed to allow organisms to reproduce and grow?

19. What is the selectivity of various herbicides to specific aquatic plants? Selective herbicides would allow other plants to grow, thus eliminating the problem of effects on all plant species. As previous discussions indicate, complete eradication of aquatic weeds is detrimental to the aquatic system.

20. Are laboratory bioassay procedures relevant to impact observed on aquatic systems? It appears to us that laboratory bioassay tests cannot predict what a chemical will do in the lake or river system.

21. Is the review process for labelling herbicides consistent with maintaining effective herbicides on the market? The cost of review is very expensive and might discourage chemical producers from developing new and more effective herbicides.

22. Are there aquatic plant growth regulators that could be applicable to the management of aquatic plants? Growth regulators are currently in use in terrestrial systems to retard plant growth. It should be determined if such regulators can be used in aquatic systems.

Mechanical Control

23. Mechanical control devices have been developed that can be used in certain situations and sites. Evaluations must be made to determine their economic and energy costs under different conditions and whether these methods are effective.

Integrated Control

24. What is the best integrative control method approach utilizing current technology? For example, how should various chemical, biological, physical, mechanical, and environmental manipulations be mixed to achieve optimal aquatic plant management?

25. Can we develop integrated mechanical, chemical, and biological control techniques that have acceptable cost-benefit ratios?

Miscellaneous

26. What are acceptable cost-benefit ratios for aquatic plant management? How should cost-benefit ratios be determined? Should public monies be spent on private water or minimize potential infestation of public waters with problem plants?

27. What are the social and economic impacts of alternative aquatic control techniques?

28. Should public monies be spent for development of control techniques? If a chemical or other technique shows promise as a beneficial method, should the research and development of this technique be funded with public monies?

Aquatic weeds can currently be controlled in most aquatic habitats with one or a combination of chemical, mechanical, biological, or habitat manipulation techniques. For the most part, the technique and extent of weed control are decided on the basis of water usage and cost. Environmental effects are often evanescent and result from removal of weeds or changing the habitat from littoral to limnetic.

The above list provides a basis for establishment of research priorities and programs. It is not intended to be complete and cover all aspects of aquatic weed research, but we feel it includes the major questions. This list, however, should form a basis for discussion that will lead to establishment of research needs and priorities and sound research programs.

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